CARBON STOCKS OF THE MANGROVE FOREST IN MWACHE CREEK, MOMBASA, KENYA

MUGI, LILIAN MWIHAKI

A thesis submitted to the Graduate School in partial fulfillment for the requirements of the Master of Science Degree in Environmental Science of Egerton University.

Egerton University

January 2015

DECLARATION AND RECOMMENDATION

I hereby declare that this thesis is my original work and has never been presented for the award of a degree in any other university and that all the sources I have used have been acknowledged.

Signature……………………………………………..…Date……………..…………………

Ms. Lilian Mwihaki Mugi

Reg No. NM12/2758/10

Approval by supervisors

Signature……………………………………………..…Date……………..…………………

Dr. George M. Ogendi

Department of Environmental Science, Egerton University,

P.O. Box 536,

Egerton.

Signature……………………………………………..…Date……………..…………………

Dr. James G. Kairo

Kenya Marine and Fisheries Research Institute,

Box 81651,

Mombasa.

Signature……………………………………………..…Date……………..…………………

Dr. Jared Bosire

Kenya Marine and Fisheries Research Institute,

Box 81651,

Mombasa.

Dedication

To Dad, Mom, Karash, Ero, Uncle Lazarus, Cucu and the entire Wairagu's family for your faith in me and the endless encouragement.

COPYRIGHT

ACKNOWLEDGEMENT

My experience in pursuing my MSc project has been challenging but enjoyable. It has been an educative venture accompanied by many hardships, frustrations, encouragement and trust and with so many people's kind and infinite help. It will not be enough to express my gratitude in words to everyone who has helped me though I would still like to extend my sincere thanks to all these people who have made the completion of this work successful.

First, I would like to extend my sincere gratitude to my supervisors who accepted the role without hesitation and offered me their wide-ranging knowledge and time. I have learnt a lot from them and without their help I could not have finished my work successfully. I am grateful to Dr. George Ogendi for his vital encouragement and support throughout my graduate studies. I also extend my gratitude to Dr. James Kairo for his unending help and inspiration coupled with the constant motivation. I acknowledge Dr. Jared Bosire, who as the principal investigator of the project, "Resilience and adaptation of mangroves and dependent communities in the WIO region to the impacts of climate change", accepted my involvement in the project, and for the support and guidance he provided throughout the tenure of my work.

I would like to acknowledge Angela, George, Eric, Angwenyi, Joyce, Sylvia and Gaya for assisting in the field sampling and laboratory analysis. My gratitude is also extended for the invaluable assistance provided in data analysis by Dr. Fulanda. Lastly, I would wish to thank the rest of my KMFRI team for support accorded to me in one way or another. The whole process would be nothing without the enthusiasm and imagination from the team.

I am highly indebted to my parents, brothers and extended family for the continuous spiritual, emotional and financial support throughout my studies. It goes without saying that my family and friends largely participated in one way or another in the success of my studies.

Successful implementation of this work was made possible through financial support from WIOMSA Marine Science for Management (MASMA) program grant no. 01003007622509; to whom I am very grateful.

ABSTRACT

It is generally accepted that escalating concentrations of atmospheric carbon dioxide (CO_2) are driving changes in climate patterns. Policy mechanisms such as 'Reducing Emissions from Deforestation and forest Degradation' (or REDD+) aim to reduce CO_2 levels in the atmosphere through compensating landowners to manage their land as carbon sinks. However, for such a scheme to succeed accurate quantification and reporting of the sequestered carbon must be conducted using verifiable methodology. Vegetated coastal habitats, such as mangrove forests, provide an opportunity to develop a carbon offset project.

In Kenya, mangroves face a myriad of human and natural induced stresses ranging from over-exploitation of resources, conversion pressure, and sea level rise. The degradation presents an opportunity for engaging in carbon markets through rehabilitation, conservation and sustainable utilization of mangrove resources. This study at Mwache creek, in Mombasa, aimed at estimating total mangrove carbon stocks in the area; in order to provide baseline information in which future offset projects could be based. Systematic stratified sampling technique was used in the study. Three carbon pools were considered, viz: Above ground, below ground (root) and soil carbon pools. Soil cores were collected at the center of 10 x 10 m^2 plots laid 100 m apart along transects. For each soil core, four sub-samples; viz., 0-15; 15-30; 30-50; and 50-100 cm were extracted for analysis of soil structure, bulk density and carbon concentration. Wet sieving was used to determine soil structure; whereas organic matter and carbon concentration were determined using loss on ignition (LOI) and the colorimetric methods. The study results indicate a statistical difference ($p<0.05$) in the vertical distribution of soil organic carbon but no statistical difference ($p>0.05$) in the horizontal distribution along the sea-land transects. A statistical difference ($p<0.05$) in the soil carbon was observed across degradation gradients with less degraded sites exhibiting higher concentrations. Above and below ground biomass was obtained using published allometric equations (230.6 and 82.7 Mg ha^{-1}, respectively) and used to determine associated carbon. The derived above and below ground carbon was added to the soil carbon to obtain total mangrove carbon of the area. The total mangrove carbon in Mwache was estimated at 388.92 Mg C ha^{-1} of which 63% was soil carbon, 28% above ground carbon, and 9% below ground carbon. These findings provide a good baseline data for establishment of a small scale blue carbon project in the area.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF ACRONYMS AND ABBREVIATIONS

AGC	Above ground carbon
BGC	Below ground carbon
CDM	Clean Development Mechanism
COP	Conference of Parties
DOC	Dissolved Organic Carbon
FAO	Food and Agriculture Organization of United Nations
GHG	Greenhouse Gases
IPCC	Intergovernmental Panel for Climate Change
IUCN	International Union for Conservation of Nature
KMFRI	Kenya Marine and Fisheries Research Institute
KPA	Kenya Ports Authority
LOI	Loss on Ignition
LULUC-F	Land use, Land use change and forestry
MASMA	Marine Science for Management Program of WIOMSA
NAMAs	Nationally Appropriate Mitigation Actions
NCCRS	National Climate Change Response Strategy
NGO	Non-governmental Organization
NIS	National Inventory Submissions
REDD+	Reducing emissions from deforestation and forest degradation
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
TEV	Total Economic Value
UNEP	United Nations Environmental Programme
UNFCCC	United Nations Framework Convention on Climate Change
USD (US$)	United States Dollar
WIO	Western Indian Ocean
WIOMSA	Western Indian Ocean Marine Scientists Association

CHAPTER ONE

INTRODUCTION

1.1 Background information

A continuous cycle of carbon between earth, atmosphere and ocean exists. There is evidence that man has largely influenced this cycle leading to increased carbon dioxide (CO_2) concentration into the atmosphere; and hence climate change (IPCC, 2007). It is estimated that tropical deforestation contributes approximately 18% emission of greenhouse gases (GHGs) into the atmosphere (IPCC, 2007); much of which is CO_2. For this reason, the role of forests in mitigating climate change effects is recognised by the Land Use and Land Use Change and Forestry (LULUC-F) sector of United Nations Framework Convention on Climate Change (UNFCCC) (Brown *et al*., 1999); as forests sequester CO_2 during the process of photosynthesis.

Carbon emission avoidance practices are encouraged to conserve existing carbon pools in forest vegetation and soil through options such as controlling deforestation or logging and other anthropogenic disturbances. A set of policies known as 'Reducing emissions from avoided deforestation and forest degradation' or REDD+ were introduced during the 11th session of the UNFCCC, in December 2005, and won support from almost all Parties, intergovernmental organizations and non-governmental organizations. REDD+ is concerned with both reducing emissions and enhancing carbon stocks through actions that address deforestation, forest degradation, forest conservation and sustainable forest management. The basic idea behind REDD+ is that countries that are willing and able to reduce emissions from deforestation and forest degradation should be compensated for doing so (Angelsen, 2008).

A key challenge for successfully implementing any REDD+ project is the reliable estimation of biomass carbon stocks in forests. Lack of information and inaccurate quantification of total sequestered carbon has made it difficult to establish the potential value of the ecosystems in global estimates and in trading of carbon credits in carbon financing programs such as REDD+. The deficiency is worse in mangrove forests owing to the logistic difficulties of working in the wetland ecosystem (Tamooh *et al.,* 2008). While several studies have been published on above ground carbon stocks in the forests around the world, there is quite limited data on below ground carbon and particularly the soil carbon (Dargusch *et al.*, 2010; Kauffman and Donato, 2012). Quantification of carbon storage in the mangroves has primarily been based on extrapolation from only a few forest surveys and inventory data (Komiyama *et al.,* 2008). The present study aimed to complement global initiatives of determining carbon stocks of coastal wetlands, commonly referred to as "Blue Carbon". The study focused on mangrove forests with an aim to provide baseline data for future engagement in carbon offset projects.

1.2 Statement of the problem

Considering the threats posed by climate change, particular interest needs to be given to cheaper ways of removing excess CO_2 from the atmosphere. Despite occupying around 2% of the seabed area, vegetated ecosystems including mangroves, seagrass beds and salt marshes transfer 50% of carbon from the ocean to sediments which mostly build up continuously while storing the carbon (Crooks *et al*., 2010). However, these ecosystem are threatened by both human and natural induced stresses including, overexploitation of resources, conversion pressure and sea-level rise. Between 1980 and 2000 for instance, 35% of mangroves were lost globally (Giri *et al*., 2011). In Kenya, losses of mangroves from 1985 to 2010 has been estimated at 18% (Kirui *et al*., 2013); with peri-urban systems of Mombasa recording up to 86% cover loss (Olagoke, 2012; Bosire *et al.,*2013). Degradation of mangroves leads to loss of ecosystem services; and discharge of previously buried carbon from the mangrove ecosystems.

Despite the potential role of mangroves as carbon sinks large uncertainties exist regarding the amount of carbon stored in the forests and particularly in their soils. Further, their variability in relation to their positioning- fringing, riverine or estuarine, basin, over-wash islands or dwarf mangroves- brings about variations in their capacity to capture and store carbon consequently leading to difficulties for a general approach in quantification. This hence calls for site-specific studies of the carbon stocks and sequestration, which would matter greatly in forest conservation and in the issues of spatial and temporal change. This study was thus undertaken to accurately quantify the Mwache Creek mangrove forest carbon stocks as a precursor for a carbon offset project for the area.

1.3 Broad objective

To assess the total organic carbon in mangroves of Mwache Creek, Mombasa; in order to provide baseline data for future engagement in carbon offset projects.

1.4 Specific objectives

i. To determine horizontal and vertical distribution of soil organic carbon along sea-land transects in Mwache Creek.
ii. To correlate soil organic carbon with mangrove degradation gradient in Mwache Creek.
iii. To use the data to estimate ecosystem carbon stocks in Mwache Creek.

1.5 Hypotheses

H_{01} There is no change in levels of soil organic carbon along the sea-land transects in Mwache Creek.

H_{02} There is no change in the quantity of soil organic carbon with an increase in depth in Mwache Creek.

H_{03} There is no variation in the quantity of soil organic carbon across a degradation gradient in Mwache Creek.

1.6 Justification of the study

Vegetated coastal ecosystems (mangroves, seagrass beds, and salt marshes) contain substantial quantities of "blue carbon" which can be released to the atmosphere when these ecosystems are degraded. For instance, mangroves contain large per-hectare carbon stocks (global stocks approximately 8 Pg C (1 Pg=10^{15} grams)) but due to their degradation they contribute approximately half the estimated total blue carbon emissions annually (0.24Pg carbon dioxide) (Donato *et al*., 2012; Pendleton *et al.,* 2012). Indications of the capabilities of mangroves as major carbon sinks are clear, setting them apart from other coastal habitats (Donato *et al*., 2012). Despite their immense values, mangroves throughout the world continue to be abused, removed and degraded (FAO, 2007). Climate change impacts further threaten the existence of mangroves from the face of the earth (Gilman *et al*., 2008). Global loss of mangroves from 1980 to 2005 reduced mangrove area by 20% (Spalding *et al*., 2010). This loss has negatively affected peoples' livelihoods, particularly communities along the coast who largely depend on mangrove products and services (IUCN, 2006).

Due to the values and the threats to mangroves, it is of interest to know the size of these carbon pools, which could lead to improvements of quantification of the global carbon stock and the sequestration capacity in different mangrove forest types. Also, in creating a baseline, carbon dynamics could determine long-term changes associated with climate change and/or land management in the mangroves (Chmura *et al*., 2003; Ray *et al*., 2011). Amid the threat of losing the ecosystem services from mangroves, an opportunity presents itself where avoiding deforestation and conservation of the carbon stocks can offer substantial benefits through climate change mitigation projects.

The present study complements previous work that aimed to determine standing biomass of mangroves of Mwache. By combining below and above ground carbon estimates, results of this study could serve as an important baseline upon which a future carbon off-set project for the area can be based along with providing an opportunity to restore the forest, ease poverty, enhance ecosystem services and also present new arguments for the conservation strategies. The study results also contribute to Kenya's REDD readiness required to support REDD implementation by providing options for REDD+ activities. The study made use of methodologies detailed in the 2013 supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands. That way, the study may be used to inform the country's National Inventory Submissions (NIS) to UNFCCC as well as providing country's options regarding Nationally Appropriate Mitigation Actions (NAMAs).

1.7 Assumptions

Due to the absence of site and species-specific allometric equations on mangroves of Kenya, generic allometric equations developed in Asia were applied in deriving above and below ground biomass. Specific wood densities developed for the mangrove in Zambezi Delta, Mozambique (Bosire *et al*., 2012) were used in the general formulae assuming similarities of the mangroves in the Western Indian Ocean region.

1.8 Scope of the study

The study was carried out in the mangrove forest of Mwache Creek. The area was chosen considering its geographical location and pressures; being a peri-urban forest where human disturbances are considerably higher compared to other remotely situated forests, and having been negatively impacted by extreme events (*El nino*). The forest was categorized into five sites depending on structure and location; KPA, Bonje, Mwakuzimu, Mashazani and Ngare. KPA represented islands within the creek which have resulted from accretion. KPA has young over-wash forest of *Sonneratia alba*. Given their location, the islands were less degraded compared to the rest of the sites. Mwakuzimu and Ngare were moderately impacted sites of mixed species stands while Mashazani and Bonje were highly impacted sites. Field sampling was done for a period of two months while laboratory analysis was carried out for a period of three months.

1.9 Limitations of the study

During the study duration, a number of limitations were encountered. There was lack of past data and a detailed vegetation map of the area that would have enabled precise temporal comparison in the biomass and carbon stock dynamics. There was also lack of an elemental analyzer for the carbon analysis which necessitated the use of a semi-quantitative method (colorimetric method) in deriving the conversion factor from organic matter to organic carbon. The absence of local factors also necessitated the adoption of specific wood densities from Mozambique, generic allometric equations from the Americas and Asia, and wood carbon concentrations from Mexico.

CHAPTER TWO

LITERATURE REVIEW

2.1 An overview of mangrove ecosystem

Mangrove forests are intertidal communities of trees and shrubs distributed in tropical and subtropical coasts around the world between 30° north and south of the equator (Tomlinson, 1986; Spalding *et al.,* 1997; Giri *et al.,* 2011). Overall, there are 15.2 million ha of mangroves around the world; down from 18.8 million ha in 1980 representing a loss of 0.18 million ha annually (FAO, 2006; Spalding *et al*., 2010). Mangroves represent less than 0.7% of tropical forests (Giri *et al*., 2011) and despite their limited area they are of global economic, environmental and social importance to humans (FAO, 1994; Costanza *et al*., 1997; Kathiresan and Bingham, 2001).

Mangroves have evolved to survive their common habitat by developing structural, morphological and reproductive adaptations that have enabled them to thrive and reproduce in harsh environmental conditions. These include exposed breathing roots, extensive support roots and buttresses, salt-extracting leaves and viviparous water-dispersed propagules (Tomlinson, 1986; Saenger and Snedaker, 1993; Kathiresan and Bingham, 2001). Mangroves vary greatly in structure and function largely as a result of topography, substrate, latitude, and hydrology (Saenger and Snedaker, 1993).

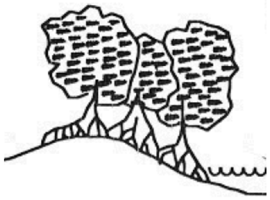
In Kenya mangroves have been estimated to cover 45,590 ha (Kirui *et al.*, 2013); representing 3% of natural forests or 1% of the state land (Wass, 1995). These forests occur in creeks, protected bays and estuaries spread along the 600km coastline from Kiunga at the Kenya Somali border to the north, to Vanga at the Kenya-Tanzania border to the south. Altogether there are 73 true mangrove species in the world (Spalding *et al*., 2010); with 9 of these in Kenya and the Western Indian Ocean (WIO) region (Spalding *et al*., 1977; Spalding *et al*., 2010). The dominant mangrove species in Kenya are *Rhizophora mucronata and Ceriops tagal* that represent more than 70% of mangrove formation (Ferguson, 1993). Other common species are *Sonneratia alba, Avicennia marina* and *Bruguiera gymnorrhiza.* Less frequent species, include;

Xylocarpus granatum, Xylocarpus mollucensis, Lumnitzera racemosa and *Heritiera litoralis* (Kokwaro, 1985; Dahdouh-Guebas *et al*., 2000).

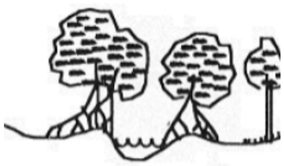
A combination of a range of biotic and environmental factors determines the distributional patterns of mangrove species and formations at global, regional, estuarine and intertidal scales. These include climate, rainfall and frequency of runoff from riverine catchments, sediment input, salinity regimes and gradient, and tides (Duke *et al*., 1998). The interactions of these factors have led to five major mangrove forest types, including; fringing, riverine/ estuarine, overwash, basin and dwarf mangroves (Lugo and Snedaker, 1974). Structural attributes of the different mangrove forest types are summarized in Box 2.1.

The peri-urban mangroves of Mwache Creek, where the current study was based, have a mixture of different mangrove types; fringing mangroves at the lower sections, riverine mangroves at the upper river-mediated sections, overwash mangroves on the forming islands within the creek, basin mangroves along depressions cross the area and dwarf mangroves towards some landward zones.

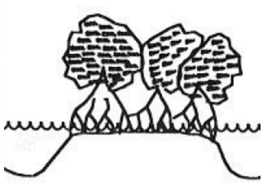
Box 2.1: Classification of mangrove forest types (*modified source:* Lugo and Snedaker, 1974)

a) 

Fringing mangroves: Tidal mediated forests found along protected coastlines and islands, and the exposed open waters. Daily tides into the forest transport nutrients into and outside of the forests.

b) 

Riverine/ estuarine forests: These occur along river and creek drainages where natural patterns of freshwater discharge remain intact. They are perhaps the most highly productive of the mangrove communities due to admixing of freshwater with seawater.

c) 

Over-wash mangroves: They occur on smaller low islands and projections in bays and estuaries, typically inundated on each tidal cycle. Unlike fringe forests, the entire island is typically inundated on each tidal cycle.

d) 

Basin forests occurring inland along drainage depressions where hypersaline conditions are likely to occur periodically due to irregular tidal action. Some of the basin mangroves like in Chale Island in Kenya have been found to be quite productive.

e) 

Dwarf forests occurring in areas where nutrients, freshwater and inundation by tides are all limited. Despite their small size and relatively low area to biomass ratios, dwarf mangroves typically have higher leaf litter production rates, thus primary production in dwarf forests is disproportionately high when compared with normal mangrove forests.

2.2 Importance of mangroves

As keystone coastal ecosystems and a source of renewable resources, mangroves are valuable for ecological, environmental and economic reasons (Kathiresan and Bingham, 2001; UNEP, 2007). Mangroves play a fundamental role in coastal protection from extreme weather conditions and natural disasters where they act as buffers, reducing vulnerability of the coasts. This was well witnessed in 2004 Asian Tsunamis in which areas with non-degraded mangroves suffered less damage than those areas that had suffered mangrove degradation and transformation (Dahdouh-Guebas *et al*., 2005; Danielsen *et al*., 2005; UNEP-WCMC, 2006). Mangroves also play a role in oxygen production, carbon retention and cycling, regulation of water quality, support of biodiversity, maintenance of breeding and rearing habitats, among others (Table 2.1). Mangroves provide many direct goods and services to millions of people along the coast; including building poles, firewood and fishery resources. It is no wonder the total economic value (TEV) of mangroves has been estimated to be ranging between 3,207 and 9,000 USD ha^{-1} yr^{-1} (UNEP-WCMC, 2006; Costanza *et al*., 1997). A study on TEV in replanted mangroves at Gazi Bay, Kenya, established the value to be USD 2902.87 ha^{-1} yr^{-1} , most of which as shoreline protection (Kairo, 2006).

Table 2.1: Outline of major ecosystem services provided by mangroves (Source: FAO, 2006)

Ecological	**Economic**	**Environmental**
Carbon retention and cycling	Commercial fisheries	Sediment trapping
Nursery grounds	Aquaculture	Coastal protection
Oxygen production	Medicinal products	Water quality regulation
Nutrient cycling	Building materials	Flood regulation
Supports high biodiversity	Salt	Land stabilization
Primary production	Tannins	
	Dyes	
	Fuel wood	
	Ecotourism & Aesthetics	

Mangroves are among the main ecological habitats along the coastal areas in the Western Indian Ocean (WIO) region (UNEP, 2012). Major uses of mangrove wood in Kenya are as building poles and fuel wood (Dahdouh-Guebas *et al*., 2000; Abuodha and Kairo, 2001). In the context of climate change, mangroves are known to capture and store large quantities of CO_2 from the atmosphere within their biomass and sediments. On a global scale, retention of carbon (allochthonous and autochthonous production) in mangroves sediments has been estimated at 385 Mg C ha^{-1} at a rate of 3.0 to 3.5 Mg C ha^{-1} yr^{-1} (Chmura *et al.,* 2003; Lovelock and Ellison, 2007). Carbon capture and storage in mangroves is dependent on forest conditions, species, age, climate, soils, topography, frequency and duration of tidal inundation (Black *et al*., 2009). In Mwache area, mangroves are depended upon by the surrounding community for a variety of direct uses including: building materials, fuel wood, fodder, and fish, among others (Ochiewo, per comm.). The present study aimed at assessing carbon stocks and sequestration potential of mangroves in a peri-urban site of Mwache creek, Kenya.

2.3 Threats to mangroves

Mangroves are among the most threatened ecosystems on earth (Valiela *et al*., 2001). Their global rate of loss has been declining since 1990 but has nevertheless remained 3-5 times faster than the overall global rate of deforestation, and with considerable variation in the rate of decline among countries (FAO, 2007). Currently, the annual decline rate of mangroves stands at 1-2 % (FAO, 2007). Globally, mangrove forests have been reduced to less than 50% of the original cover (Spalding *et al*., 1997) with a 25% decline between 1980 and 2000 according to FAO (2006). In the WIO region the loss of mangroves between 1980 and 2005 has been estimated at 8% (FAO, 2007a). In Kenya, there was a 0.7% annual loss between 1985 and 2010 (Kirui *et al*., 2013).

On a global scale, major threats of mangrove forests have been ranked as over-exploitation of wood products, conversion of mangrove areas to other land uses (such as aquaculture, saltpans, agriculture and human settlement), diversion of freshwater flow and mining, pollution and damming of rivers that alter water salinity (Alongi, 2002; Nguyen, 2005; FAO, 2007b; Giri *et al*., 2008). Oil spills have impacted mangroves along the coasts of Africa and in the Caribbean (FAO, 2007b).

Perhaps the major threat to mangroves in the world today is the conversion to other land uses. 20 to 50% of mangroves loss worldwide resulted from conversion to ponds for shrimp aquaculture (Primavera, 1997; Valiela *et al*., 2001). In the Indo-Western Pacific region alone, 2.0 million hectares of mangroves had been converted to aquaculture ponds by 2000 (Primavera, 2005). In the WIO region, loss of mangroves to aquaculture is not extensive. However, localized clearing of mangrove for pond culture has destroyed mangroves of Ngomeni in Kenya, Tanga and Rufiji in Tanzania, as well as north east part of Nosy Be in Madagascar (UNEP 2003). In Kenya, 100 hectares at Ngomeni were converted to pond culture in the 1980s and later abandoned after the trial exercise.

Mangrove forests are the principal sources of wood products for building and fuelwood in coastal areas of Africa and South East Asia (FAO, 2007). In Kenya, mangroves provide up to 70% of wood requirement to the adjacent communities (Wass, 1995; FAO, 2007). Over harvesting has led to resource depletion, decline in fisheries and increased shoreline erosion. The situation is worse in peri-urban mangrove areas such as the Mwache mangrove forest which are under pressure due to over-harvesting for domestic firewood by the populace (Abuodha and Kairo, 2001).

The location of mangroves in the land-sea interface makes them quite vulnerable to the effects of climate change particularly, the sea level rise (Lovelock and Ellison, 2007). According to the IPCC (2007) predictions sea-level rise in the eastern Africa region has been predicted to be almost 70 cm by 2100. Such a rise in sea level will submerge low lying coastal areas killing mangroves (GoK, 2009). Under natural conditions mangroves have the ability to keep pace with changing sea-levels if the rate of change in elevation of the mangrove sediment surface exceeds the rate of change in relative sea-level, and if there is adequate space for the expansion (Field, 1995; Mcleod and Salm, 2006). Growth of mangroves and their areal extent may be affected by changes in precipitation patterns caused by climate change (Field, 1995). The 1997/98 *El-Nino* rains along the Kenya coast increased sediment loading into mangroves of Mwache creek and other areas; smothering the root systems of trees and causing die-back of the forest (Kitheka *et al.* 2003; Kitheka, *et al.* 2005).

The peri-urban mangroves of Mwache Creek are under direct pressure from over-extraction, conversion and pollution due to their location (Bosire *et al*., 2013). The present study aimed at determining the carbon stocks in the mangroves of Mwache Creek as well as provided a baseline for projections and estimations of emissions/ additions with time. In addition, emissions from changes in forest cover were estimated using default IPCC values (IPCC, 2014).

Blue carbon habitat losses result to roughly 58,000 Mg C annually, with about sixty percent of this coming from conversion of mangroves to other land uses (Siikimaki *et al*., 2012a). Mitigating against mangrove carbon emissions has economic potential estimated at USD4 to USD10 per tonne of CO_2. This varies with location, emissions, costs of avoiding the emissions and risks associates with emissions (Duarte *et al*., 2005; Siikimaki *et al*., 2012a; Siikimaki *et al*., 2012b).

2.4 Forests and climate change

There is an unequivocal relationship between forests and climate change as the latter has affected forests and their ability to play their functions. On the other hand, degradation of the natural resources has resulted to emissions of CO_2 in the atmosphere (Lasco *et al*., 2008). Past and present patterns of land use are responsible for the current situation in regard to the carbon pools and changes of the world's forests. From the late 1800s until about the 1930s, global CO_2 emissions from changes in land use were similar in magnitude to those from fossil fuel combustion. Around the 1950s, CO_2 emissions from changes in forest use in the tropics dominated the releases to the atmosphere, worldwide fossil fuel use soared, biotic emissions from the mid and high latitude regions declined greatly as forests expanded onto abandoned agricultural lands as logged stands regrew, and deforestation in the tropics accelerated (Brown, 1999).

On a global scale forests are still estimated to be a source of atmospheric CO_2 mainly because of deforestation and forest degradation in the tropics (Brown *et al*., 1996). In addition to the many human-induced pressures, climate change is creating an additional pressure that could change or endanger these ecosystems (IPCC, 2007). As a major carbon sink, forests store in excess of 830 Pg ha^{-1} (Brown, 1998); this is more carbon per unit area than any other vegetation type (Houghton, 2007; Dargusch *et al.*, 2010). Temperate and boreal forests have been appraised as carbon sinks because many are recovering from past disturbances and they are actively managed (IPCC, 2007; FAO, 2007). Regrettably, tropical deforestation accounts for 18% of GHG emissions into the atmosphere (IPCC, 2007). In Africa, deforestation and forest degradation accounts for nearly 70% of its total GHG emissions (FAO, 2006).

The United Nations Framework Convention on Climate Change (UNFCCC) and the Kyoto Protocol were negotiated by the global community with an aim of stopping and reversing the GHG concentrations in the atmosphere. The two treaties provide a negotiation platform, an institutional framework and the technical infrastructure necessary to define international solutions to climate change. Through the IPCC, UNFCCC provides for intergovernmental actions such as sharing information on GHG emissions and developing strategies for addressing and adapting to climate change (Charlotte and Scholz, 2006; UNFCCC, 2008).

A fundamental milestone was achieved at Conference of Parties (COP) 11 in Montreal in 2005 where REDD was proposed (Parker *et al*., 2009). Its scope expanded to REDD+ indicating enhancement of degraded forest and reforestation activities and in 2010 at the 16th COP in Cancun, it was formally adopted. 'Avoided deforestation', including emissions reductions from tropical deforestation projects, which were previously excluded have gained importance in future climate change policies (Gibbs *et al.,* 2007). The REDD+ concept proposed the provision of financial incentives to help developing countries voluntarily reduce national deforestation rates and associated carbon emissions below a baseline. It is considered a mechanism for achieving the global emissions reduction targets (IPCC, 2007); those that demonstrate emissions reductions, which could combat climate change, conserve biodiversity and protect other ecosystem goods and services, may be able to sell those carbon credits on the international carbon market (Scholz and Schmidt, 2008). The global carbon market consists of an obligatory market (regulated by the Kyoto protocol under UNFCCC) and a voluntary market.

As a signatory to Kyoto Protocol and UNFCCC, Kenya is expected to benefit from the prospects of carbon trade in 'Land use and Land-Use Change and Forestry' (LULUCF) sector through REDD+ and CDM. Kenya has also developed the National Climate Change Response Strategy (GoK, 2010), which seeks to strengthen nationwide focused actions towards adapting to, and mitigating against a changing climate by ensuring commitment and engagement of all stakeholders while taking into account the vulnerable nature of our natural resources and society as a whole. Kenya is well on its way to eligibility for REDD+ funding guided by the National REDD+ Steering Committee under six thematic areas with three already operational (REDD Readiness Progress Fact Sheet Kenya, 2013).

2.4.1 Mangroves and climate change

Mangroves were previously not included in REDD until its expansion to REDD+ (Climate focus, 2011). The carbon sinking potential of mangroves and other vegetated coastal habitats are receiving heightened interests in climate change mitigation and adaptations. Mangroves and other vegetated coastal habitats (commonly referred to as 'blue forests') are among the major carbon sinks in the world with potential to mitigate climate change (Chmura *et al.,* 2003; Bouillon *et al*., 2008; Nellemann *et al*., 2009; Tue *et al.,* 2011; Mitra *et al*., 2011*)*. Under the blue carbon concept, mangroves have been estimated to sequester 3-4 times more carbon than any productive terrestrial ecosystem (Kauffman and Donato, 2012), which is captured in the below and above ground components; while a bigger part (50 to 90%) is captured by the mangrove sediments (Bouillon *et al.*, 2008; Kauffman *et al.,* 2011; Ray *et al*., 2011; Donato *et al*., 2012). Sediment carbon could either be produced *in situ* or captured by the efficient particle-trapping mechanism of mangroves made up of complex root structures. Sediment accretion in mangroves has been estimated to sequester 10 times more carbon than that observed in temperate forests and 50% more than other tropical forests (Figure 2.3). Globally, the buried carbon in mangroves has been estimated at 18.4 Tg Cyr^{-1}(1 Tg=10^{12} grams) (Laffoley and Grimsditch, 2009), approximately 8% of the annual total oceanic burial of organic carbon (Duarte *et al*., 2005). With this ability, mangroves are good candidates in carbon markets, in which carbon payments do not depend on the size of the carbon stock but rather on the carbon sequestration rate (Alongi, 2011). On the downside, degradation of mangroves leads to equally high emissions.

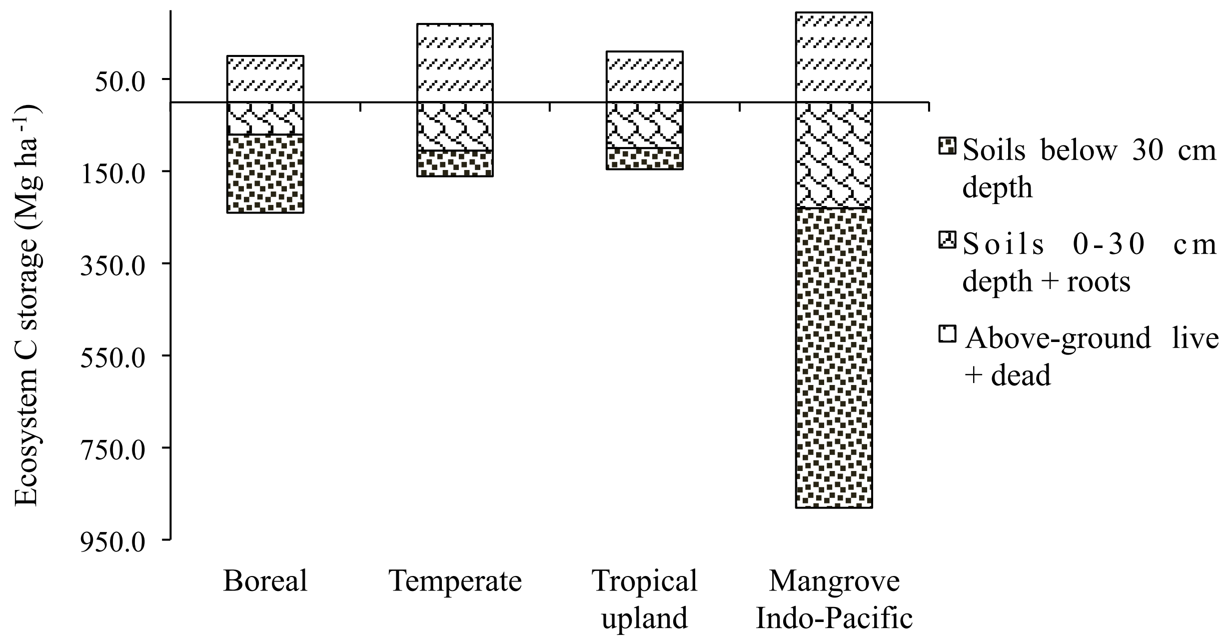


Figure 2.1: Default IPCC Values of ecosystem C pools for some major land cover types on Earth. (Modified source: Kauffman and Donato, 2012).

The present study sought to assess the carbon stocks in the different pools in Mwache Creek mangrove forest. A baseline of the carbon stock of the area was established, which may be used for carbon dynamics in the area and in developing carbon offset project for the area.

2.5 Methodological development in carbon inventory

The assessment of carbon stocks in coastal wetlands including mangroves is complicated by lack of sufficient data and methodology (Donato *et al.*, 2012). This is further complicated by variability of growing conditions of these habitats where they are influenced by a wide array of environmental variables. The 2006 IPCC guidelines on national GHG Inventories provide comprehensive monitoring and reporting of GHG on LULUC-F. These, however, do not provide specific guidance for the estimation and reporting of anthropogenic GHG emissions from and by removal of mangroves. The recently adopted 2013 supplement on 2006 IPCC guidelines on wetlands addresses this omission (Herr *et al.*, 2011).

In the present study the protocol by Kauffman and Donato (2012) was adopted in measuring the carbon stocks in the mangroves of Mwache Creek. The same protocol has been used in carbon assessment for mangrove forests in Micronesia, Mozambique, Madagascar and Kenya, making the study comparable. The above ground (live trees), below ground (root), dead wood, litter and soil are the major carbon pools considered by Kauffman and Donato (2012). The approaches are the best so far for mangrove ecosystems and are in consistence with IPCC guidelines and relevant sourcebooks and provide information on field measurements and computations that would support entry into regulatory or voluntary carbon markets.

CHAPTER THREE

MATERIALS AND METHODS

3.1 Description of the study site

3.1.1 Site and forest structure

The study was conducted at Mwache Creek (04°3.01´S & 39.06°38.06´E) (Figure 3.1), located on the upper part of the Port Reitz, 20 km Northwest of Mombasa city (Kitheka, 2000). The total area of Port Reitz Creek is estimated at 1,009 ha with about 64% of the surface area being covered with mangroves (Kitheka, 2000; Bosire, 2010). The fringing mangrove forest of Mwache creek is river mediated and contains five species of the nine described mangrove species in Kenya (Kaino, 2013). These species display a horizontal zonation pattern typical of other mangrove areas in Kenya (Kairo *et al*, 2001; Kaino, 2013). The seaward side is occupied by *Sonneratia-Rhizophora-Avicennia* community; this is followed by *Rhizophora-Ceriops* community in the mid zone and dwarf *Avicennia* on the landward side. *Bruguiera gymnorrhiza* is also present but rare within the forest. Mangrove associates such as *Sueada maritima* and the grass species *Sporobolus spicatus* were observed in the land-ward and degraded areas.

Five sites in Mwache Creek were studied, namely; KPA, Mashazani, Ngare, Mwakuzimu and Bonje. KPA, an overwash mangrove forest, represented a young monoculture stand of *S. alba* on islands not colonized before 1992 (Fergurson; 1995). The islands had young, high quality form and high stand density (Kaino, 2013). Mashazani, located on the upper reaches of the creek, was considered a highly impacted site with total destruction evident from the old and recent massive cuttings due to its easily accessible location. This site was characterized by high silt deposition, evident from shallow sandy soils and the presence of a large sandflat adjacent to the forest, from adjacent mainland where a lot of farming activities were taking place. Most of the trees in this site were stunted *C. tagal* and *R. mucronata*. Mwakuzimu, located on the lower reaches and towards the mouth of the Creek, recorded all the five mangrove species in Mwache Creek. It had tree heights spread across the various size classes with a high structural complexity. Ngare had the highest structural complexity, with trees with the highest basal area and height in the study site. It was located close the Creek's mouth and across the Mkupe jetty. Bonje was located at the uppermost reaches of the Creek. It was heavily impacted on by sedimentation from the 1997/98 *El nino* rains (Kitheka et al., 2002; Bosire, 2010).

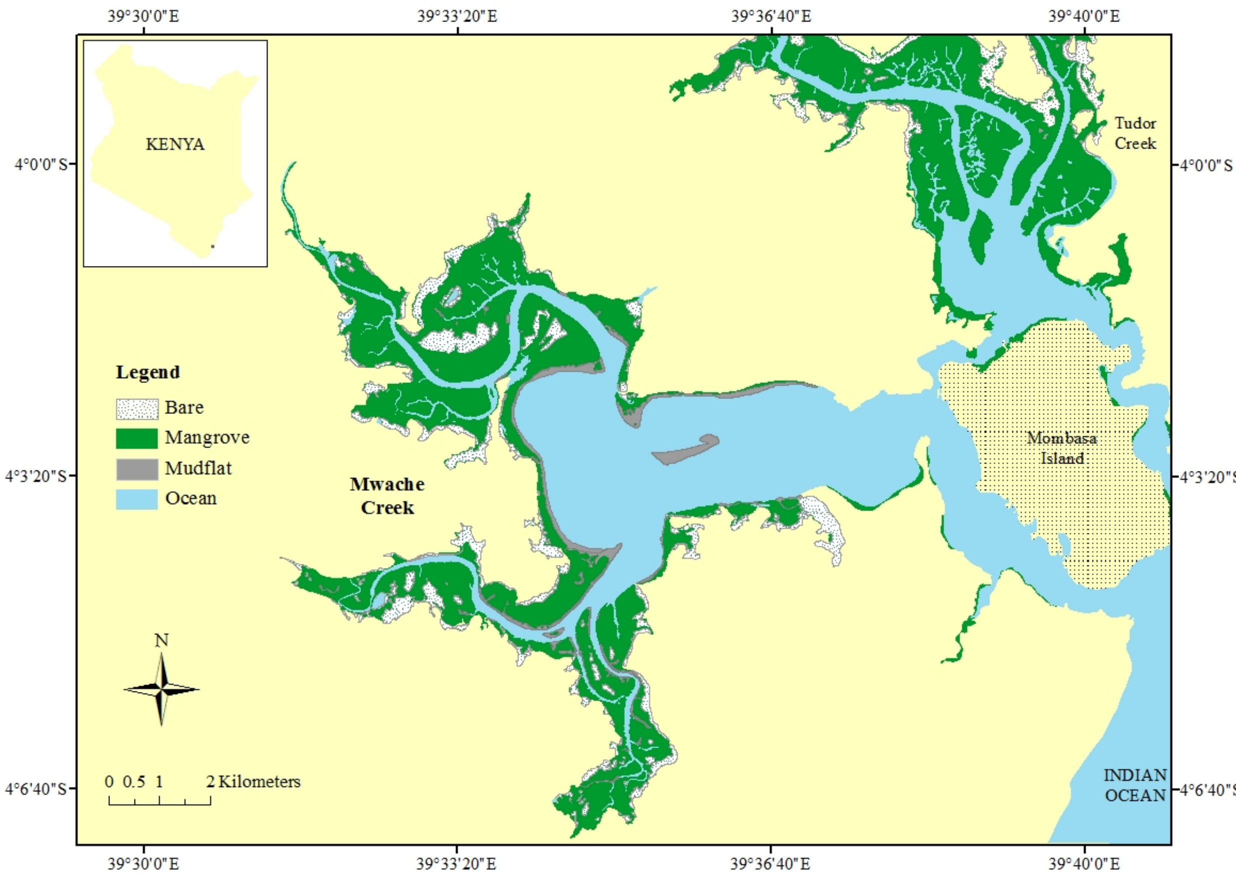


Figure 3.1: A map of study area showing mangroves in Mwache Creek

3.1.2 Climate

The coastal climate in Kenya is influenced by monsoon winds with two rainy seasons. Heavy rains occur during the South Eastern Monsoon between March and May; while short rains fall during North Eastern Monsoon from October to November. Total annual rainfall shows great inter-annual variability with mean values in the order of 900 mm. August/September and January/February are usually dry. Temperature at the coast of Kenya ranges between 24˚C.and 32.5˚C. The highest temperatures of 28-29˚C occur following the Northeast Monsoon in the months of March and April. Annual evaporation is around 1800 mm and this is considered to be higher than the normal annual total rainfall and thus a freshwater deficit in dry seasons in the basin. Evaporation increases from a low of 138 mm in July to a high of 221 mm in March. Relative humidity is comparatively high all year round, reaching its peak during the wet months of April to July.

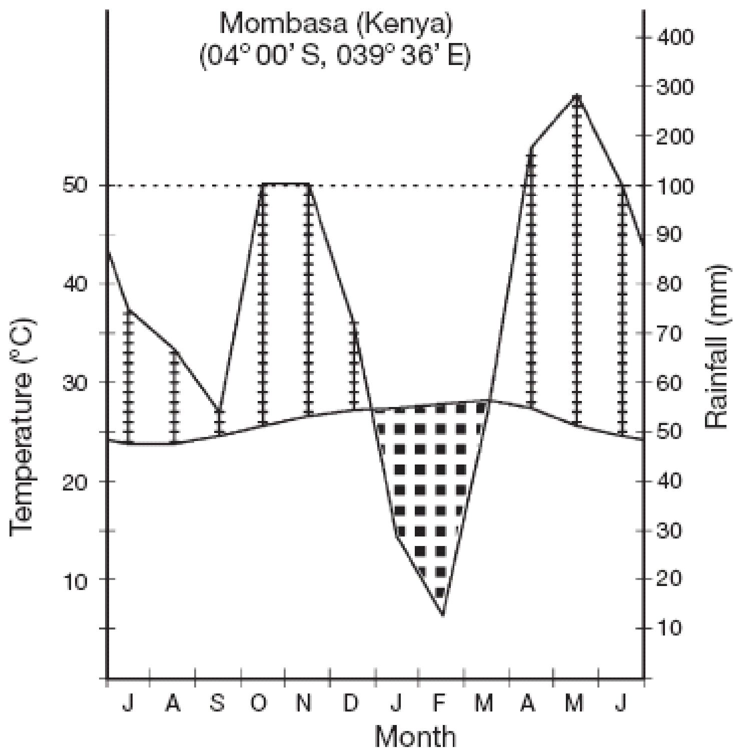


Figure 3.2: Temperature and rainfall of Mombasa (Lower continuous line is temperature, upper continuous line is rainfall (Source: Dahdouh-Guebas *et al.,* 2004)

3.1.3 Socio-economic Activities

Based on the 2009 census, there are about 6,226 households in sub-locations adjacent to the mangrove forest, with an estimated population of 36,614 (population density = 200) (GoK, 2010). Important sources of livelihood in the area are similar to rural coastal areas in Kenya (Ochiewo, 2001); Agriculture and fishing are the major activities of the people in the area. Fishing and mangrove harvesting activities are mostly associated with communities residing around the creek. Erratic rainfall in the area has largely affected farming activities with most of the community living below the poverty line (Ochiewo, per. comm.). Illiteracy levels in the area are as high as 90% (Ochiewo, per. comm.). In terms of infrastructure, the area lacks sufficient basic amenities; the access roads are usually inaccessible during the rainy season while majority of the households lack electricity thus depend on biomass fuel as a source of energy. Housing in the area is made up of simple frames of mangroves and clay. Small-scale quarry mining is the major industrial activity in the area.

3.1.4 Some human induced stresses on marine environment in Mwache

Poor land use practices in the hinterland have increased sediment loads into the mangrove forest leading to increased sedimentation and death of mangroves (Kitheka *et al*., 2002). The rate of sediment production within Mwache River basin reaches a high of 3,000 tons per year due to poor land-use activities upstream such as overgrazing, shifting cultivation, high rainfall intensity during the rainy season and steep land gradient (Kitheka *et al*., 2002; Bosire *et al*., 2006). Oil pollution from the Mombasa Port and surrounding waters further cause degradation of Port Reitz creeks. Between 1983 and 1993 Mombasa port and surrounding waters experienced 39 680 tonnes of oil spills that affected mangroves of Port Reitz and Makupa creeks. The most recent was experienced in 2005 and affected 234 ha of mangroves in the creek (Kairo *et al*., 2005). Spot assessments in some impacted site within Mwache Creek area have indicated limited post-impact recovery of the mangroves after the *El-nino* event in 1997/98 (Kitheka *et al.,* 2002; Bosire *et al*., 2008; Kaino, 2013). Clearing of mangrove trees to create access routes to shorelines and pave way for physical developments is another challenge faced in the area with plans underway for the construction of a by-pass through the area. This may cause changes in hydrology and encourage erosion of the shoreline, which maybe subsided through compensation by afforestation at another site or restoration of other degraded areas.

3.2 Sampling design

The study involved estimation of total carbon pools in Mwache. Both above and below ground biomass was estimated; from which were derived vegetation carbon (Figure 3.3). To enable accurate measurement of the carbon stock in the forest three carbon pools were considered: above ground, below ground and soil carbon pools. Data on vegetation carbon was pooled from the sister study in the same area by Kaino (2013). Although IPCC recommends accounting of litter and deadwood carbon, their contribution in the present study were deemed minimal because of faunal activity and collection of firewood by local community.

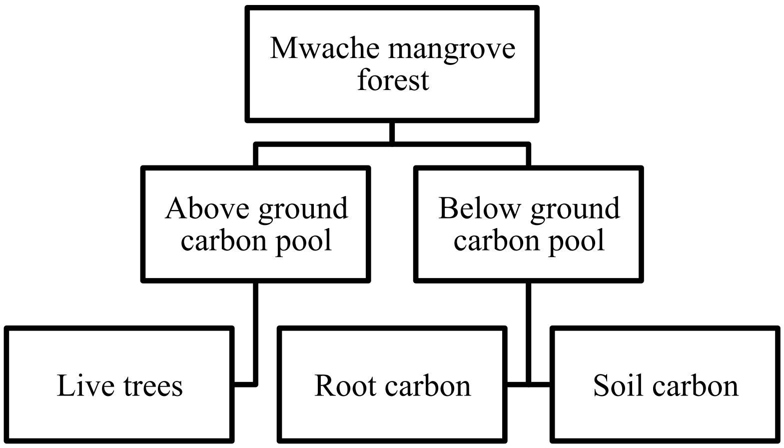


Figure 3.3: Study framework

Stratified systematic sampling was adopted based on high resolution SPOTX imagery. Species composition, extent of intertidal area, stand structure and level of degradation was taken into account where the sampling was stratified across a broad range of stand conditions to ensure representative sampling. Transect perpendicular to the water line were used.

For further comparison across a degradation gradient, one site was divided into highly degraded and less degraded sites; Bonje A and Bonje B, respectively. Bonje, in general, was considered as the degradation gradient owing to massive die backs experienced from the impacts of the 1997/98 and 2006 *El Nino* events. Bonje A was worse hit (over 80% loss) in comparison to Bonje B which had also been exhibiting regeneration over time.

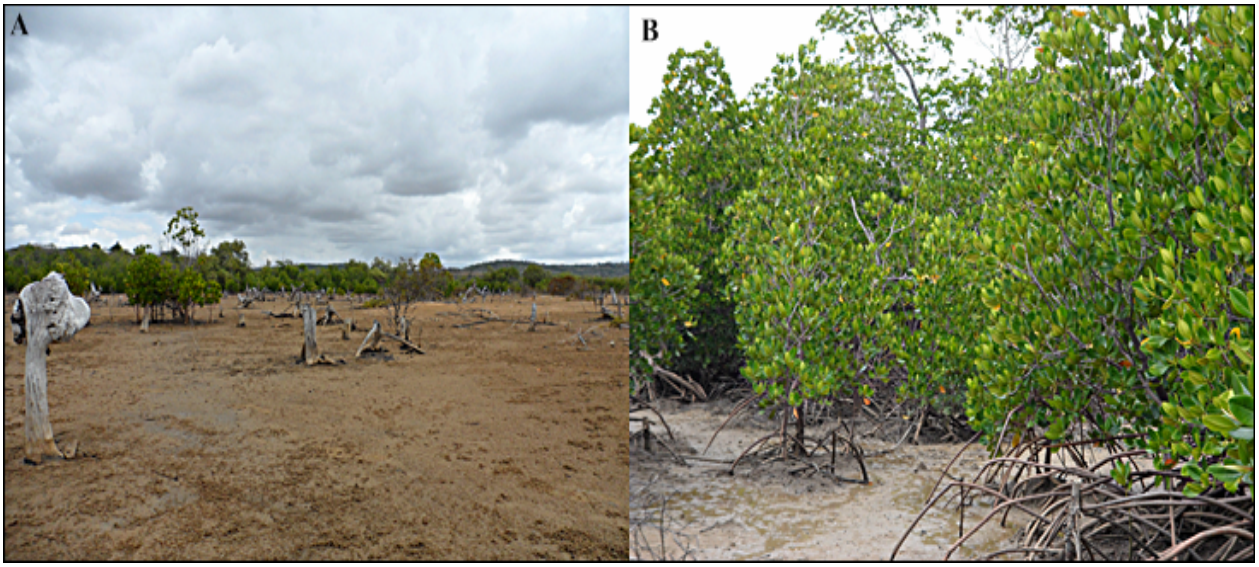


Figure 3.4: Photo of Bonje A (highly degraded) and Bonje B (less degraded) sites in the mangrove forest, Mwache Creek (Photo by: Mwihaki L., 2011)

3.3 Assessment of forest structure

In the vegetation surveys by Kaino (2013), 10m x 10m plots were laid along transects. All trees with diameter greater than 2.5 cm were identified and height (m) and diameter (cm) taken. Tree height was measured using a graduated pole, while DBH was measured using forest calipers. The density of trees per plot was also recorded. Above and below ground biomass were generated from the inventory data using general allometric equations. Kaino (2013) also determined parameters such as basal area, relative density, relative dominance and relative frequency to further describe the structure of the mangrove stand.

3.4 Soil sampling

Soil sampling was undertaken during low-tide along the sea-land transect. Soil cores were extracted from the center of each of the plots using a 4cm half-arc soil core sampler. The sampling points were again marked using a GPS. Sample contamination was prevented by washing of the corer and wiping of sub-sampling tools with each use.

The soil profile was systematically sub-sectioned into 0-15, 15-30, 30-50 and 50-100 cm which has been suggested as most appropriate partitioning (Kauffman and Donato, 2012). Similar protocol has been applied in studies in Indo-Pacific mangroves forests, Mozambique, Madagascar and other sites in Kenya. Sub-samples of around 5 cm length collected at the approximate mid-point of each depth interval and comprising at least 30-50 g of sample mass were taken. The sub-samples were sealed, labeled, stored at 4.0˚C and transported to the laboratory for further analysis.

3.4.1 Laboratory Analysis

To appropriately quantify soil carbon stocks, two parameters were measured: bulk density and organic carbon concentration. Grain size distribution was also analyzed as it influences soil organic matter / carbon distribution.

3.4.1.1 Grain size analysis

Grain size analysis was carried out to determine the fractions of silt and clay, fine and medium size sand and course sand. Samples were dried at 60˚C for 48 hours. 25g of the sample was weighed out, placed in a labeled beaker with 250ml water and 10ml of aqueous sodium hexametaphosphate (6.2g/l dilution) added to separate the soil particles. This was stirred for 10 minutes and left to settle for a minimum of four hours, after which it was stirred again for 10 minutes. The contents of the beaker were then poured into a 63µm sieve and flushed with water while brushing until no further silt was lost. The remnants were carefully brushed into a marked and pre-weighed petri dish and left to dry in the oven for 8 hours. These were then passed through a 500µm sieve stacked on a pan, after which they were separately weighed and recorded appropriately.

3.4.1.2 Bulk Density analysis

Bulk density of soil refers to the dry weight of soil per unit volume; it is the indicator of soil compaction. In the laboratory, samples for bulk density analysis were placed on pre-weighed crucibles and oven-dried to a constant mass at 60˚C after which they were weighed. The bulk density was calculated using equation (1):

Soil bulk density (gm^{-3}) =Oven-dry sample mass (g) / Sample volume (m^3)………. **Equation – 1**

Where,

Volume = Cross-sectional areas of the corer × the height of the sub-sample...........**Equation - 2**

3.4.1.3 Soil Organic Carbon Analysis

a) Loss on ignition

Soil organic matter (SOM) was determined using loss-on-ignition (LOI), a semi-quantitative method based upon the indiscriminant removal of all organic matter. For maximum efficiency, samples analysed for bulk density were used. The oven-dried samples were homogenized by grinding to a fine powder using a mortar and pestle, passed through a 2 mm sieve and placed in pre-weighed aluminum crucibles. These were set in a muffle furnace for combustion at 450˚C for 8 hours, after which they were cooled in a dessicator and weighed. Organic matter content was determined as;

{Initial weight (g) – Final weight (g) }/ Initial weight (g) ×100 ……..…………....**Equation - 3**

b) Colorimetric method

Samples were analysed for total organic carbon using the colorimetric quantitative method at the National Agricultural Research Laboratory (NARL) in Nairobi. The results were expressed as the percentage SOC per sample (%C).

3.4.2 Calculation of the total organic carbon

Data from the study on the vegetation structure was used to estimate the above ground (AG) biomass using the general equation by Komiyama *et al*. (2005). Regional species specific wood density data (Bosire *et al*., 2012) in calculating the biomass:

$AGB = 0.251\rho D^{2.46}$……………………………………………………**Equation - 4**

Where,

AGB= Tree AG biomass (kg)

ρ= wood density (g/cm^3)

D= tree diameter at breast height (cm)

The AG carbon pool was derived by multiplying the biomass of individual component tree species by their specific wood carbon concentrations, 47.1% for *S. alba*, 46.3% for *B. gymnorrhiza* and 46.4% for all other species (Kauffman *et al*., 2011).

Below ground root biomass was derived from a generalized equation by Komiyama *et al.* (2008):

$BGB = 0.199\rho^{0.899} \times (D)^{2.22}$……………….……………………………..…**Equation - 5**

Where,

BGB= Tree BG biomass (kg)

ρ= wood density (g/cm^3)

D= tree diameter at breast height (cm)

The C stock in the BG biomass was calculated as the product of BG biomass and C concentration where a default value 39% was used as the BG biomass C concentration (Kauffman and Donato, 2012), as illustrated:

$BGC = BGB \times 0.39$………………..…………..………………...…………………**Equation - 6**

To determine the total soil carbon pools after were determined by summing the mass of each sampled soil depth. The soil carbon mass per sampled depth interval was calculated as:

SOC (Mgha-1) = Bulk density (g cm-3) × Soil depth interval (cm) × %C ……….…..**Equation - 7***i*

where %C was expressed as a whole number.

The total soil carbon pool was equal to the sum of the carbon mass of the soil depths.

The results were scaled to per-hectare basis for ease of comparisons. Total C stock in the mangrove forest at Mwache creek was calculated as:

$Total\ C\ stock\ (MgCha^{-1}) = AGC + BGC\ + SOC$……………………..………………**Equation – 8**

3.5 Data Analysis

Using EXCEL and STATISTICA, graphical presentation and descriptive analyses of the data were carried out. Normality test and homogeneity of variance was done using Kolmogrov-Smirnov test and Levene's test, respectively. Data that met the normality assumption was further analysed for any significant differences using ANOVA. Data that failed to meet the assumptions of normality and homogeneity of variance were analysed using non-parametric tests; Kruskal-Wallis ANOVA and Mann-Whitney U tests. The spatial variations in organic matter, soil C concentration, grain-size distribution and bulk density were determined using the aforementioned tests. Relative standard errors were determined to avert any errors. A 95% confidence interval (CI) was used to reflect the degree of precision in the dataset. Using representative samples, a simple linear regression was developed between soil organic matter from loss on ignition (LOI) and the soil organic carbon concentration obtained through the colorimetric method.

CHAPTER FOUR

RESULTS

4.1 Vegetation Carbon

4.1.1 Above ground carbon

Based on inventory data provided by Kaino (2013) (Table 4.1), vegetation carbon pools were estimated. The above ground live biomass (AGB) in the mangrove forest of Mwache Creek was 230.6 Mg ha^{-1} (range: 175.0 to 335.9 Mg ha^{-1}). From this study, the average above ground carbon (AGC) in the area was estimated to be 107.5±14.8 Mg C ha^{-1} (range: 82.4 to 157.0 Mg C ha^{-1}) (Table 4.2). This represented 27.7% of the organic carbon stock in the Mwache mangrove forest.

Table 4.1: Summary of the vegetation density (stems ha^{-1}) ± SE and biomass (Mg ha $^{-1}$) ± SE distribution in the mangroves of Mwache Creek

Site	Tree density (stems ha^{-1})	AGB (Mg ha^{-1})	BGB (Mg ha^{-1})	Total biomass (Mg ha^{-1})
Maguzoni	2633±44	181.9±68.5	72.0±24.5	**253.9±92.9**
Mashazani	84±9	187.7±34.6	74.9±12.5	**262.5±47.0**
Mwakuzimu	1840±22	272.7±43.2	106.3±15.8	**379.4±59.0**
Ngare	1448±18	335.9±51.7	130.3±18.9	**466.2±70.1**
KPA	2000	175.0±40.1	73.9±14.1	**245.0±54.1**
Average	**1691**	**230.6±31.7**	**92.5±12.4**	**323.1±44.1**

4.1.2 Below ground carbon

Below ground (BG) root biomass was estimated at 92.5 Mg C ha^{-1} (range: 73.9 to 130.3 Mg ha^{-1}) (Table 4.1). Using C concentration of 39% of the BGB, the below ground carbon

(BGC) was 35.2±4.3 Mg C ha^{-1} (Table 4.2), representing 9.1% of the total organic carbon in Mwache mangrove forest.

The mean total vegetation carbon was 142.8±19.1 Mg C ha^{-1}, ranging from 111.3 to 206.4 Mg C ha^{-1} (Table 4.2).

Table 4.2: Summary of the vegetation organic carbon (Mg C ha^{-1}) ± SE distribution in the mangroves of Mwache Creek.

Site	AGC (Mg C ha^{-1})	BGC (Mg C ha^{-1})	Total biomass C (Mg C ha^{-1})
Maguzoni	84.4±31.8	28.1±9.5	112.5±41.3
Mashazani	87.3±16.1	28.5±4.7	115.7±20.8
Mwakuzimu	126.6±20.0	41.4±6.2	168.0±26.2
Ngare	157.0±24.3	49.3±7.4	206.4±31.7
KPA	82.4±18.9	28.8±5.5	111.3±24.3
Average	**107.5±14.8**	**35.2±4.3**	**142.8±19.1**

4.2 Soil carbon

4.2.1 Soil structure

Soil structure was expressed as percentage silt and clay (<63 µm particular sizes). The sand proportion was ignored in this study given that it was negligible in the soils in the study sites. The average percentage silt and clay was highest at Mashazani and lowest at Bonje A, accounting for 90.0±1.5 and 47.0±18.9% of soils, respectively, and averaged at 71.0±2.2% in the mangrove forest of Mwache Creek (Table 4.3). There was a significant difference ($H=44.2$; $p<0.05$) observed among all the sampled sites.

Table 4.3: Mean ± SE percentage silt and clay in the different depth intervals of Mwache sites

Depth interval (cm)	Site						
	Mashazani	KPA	Ngare	Mwakuzimu	Bonje A	Bonje B	Average
0-15	91.2±1.8	53.4±1.4	81.1±8.9	81.9±6.7	27.1±8.3	70.0±6.1	**64.8±5.6**
15-30	89.2 ±3.2	58.9±2.2	70.2±13.4	79.2±9.0	44.1±7.0	84.6±5.2	**71.0±4.6**
30-50	89.3±3.7	57.5±0.2	73.6±11.9	81.2±9.5	57.8±3.0	81.7±6.6	**73.9±3.6**
50-100	90.3±4.6	65.6±3.5	74.7±11.7	71.5±12.4	58.9±6.8	82.3±4.5	**74.1±3.6**
Average	**90.0±1.5**	**58.9±1.9**	**74.9±5.1**	**78.5±4.3**	**47.0±4.2**	**79.7±2.9**	

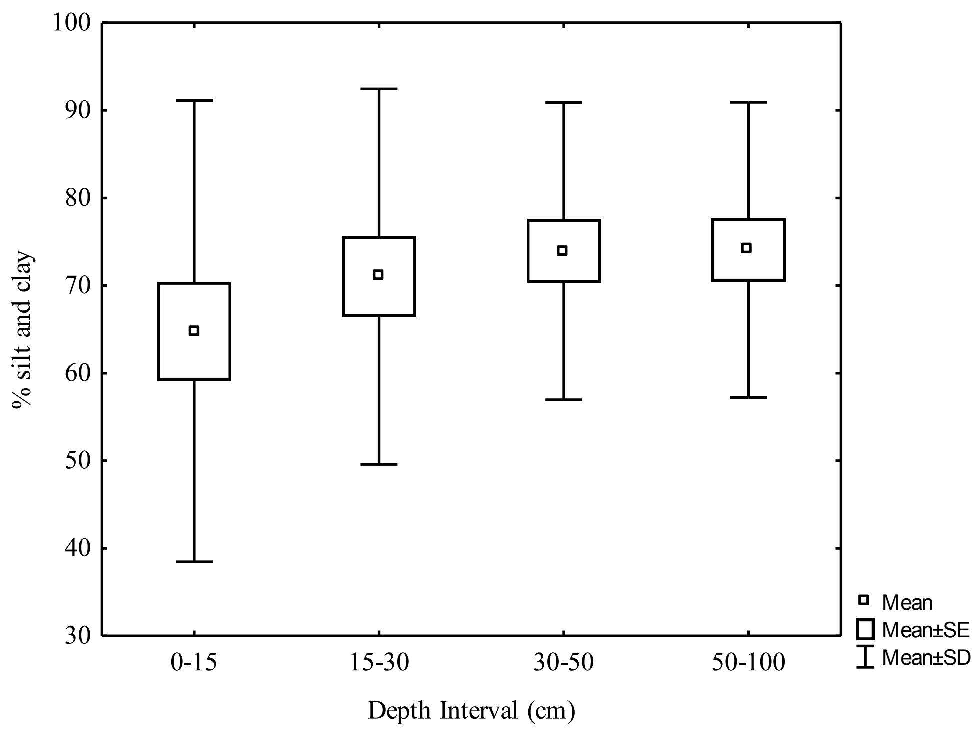


Figure 4.1: Box plot of the percentage silt and clay of the mangrove soils with depth in Mwache Creek.

Though there was an increase in the percentage silt and clay with an increase in depth (Figure 4.1), there was no significant difference in the values in the different depth intervals (H=1.2; p>0.05). The 0-15 depth interval had 64.8±26.3% while the 50-100 depth had 74.1±16.8 % of silt and clay. There was no statistical difference (H=5.4; p>0.05) in soil structure along transects, where the sea-ward, mid-tidal and land-ward zones represented 72.7±2.6, 70.5±3.9 and 80.9±4.1%, respectively (Figure 4.2).

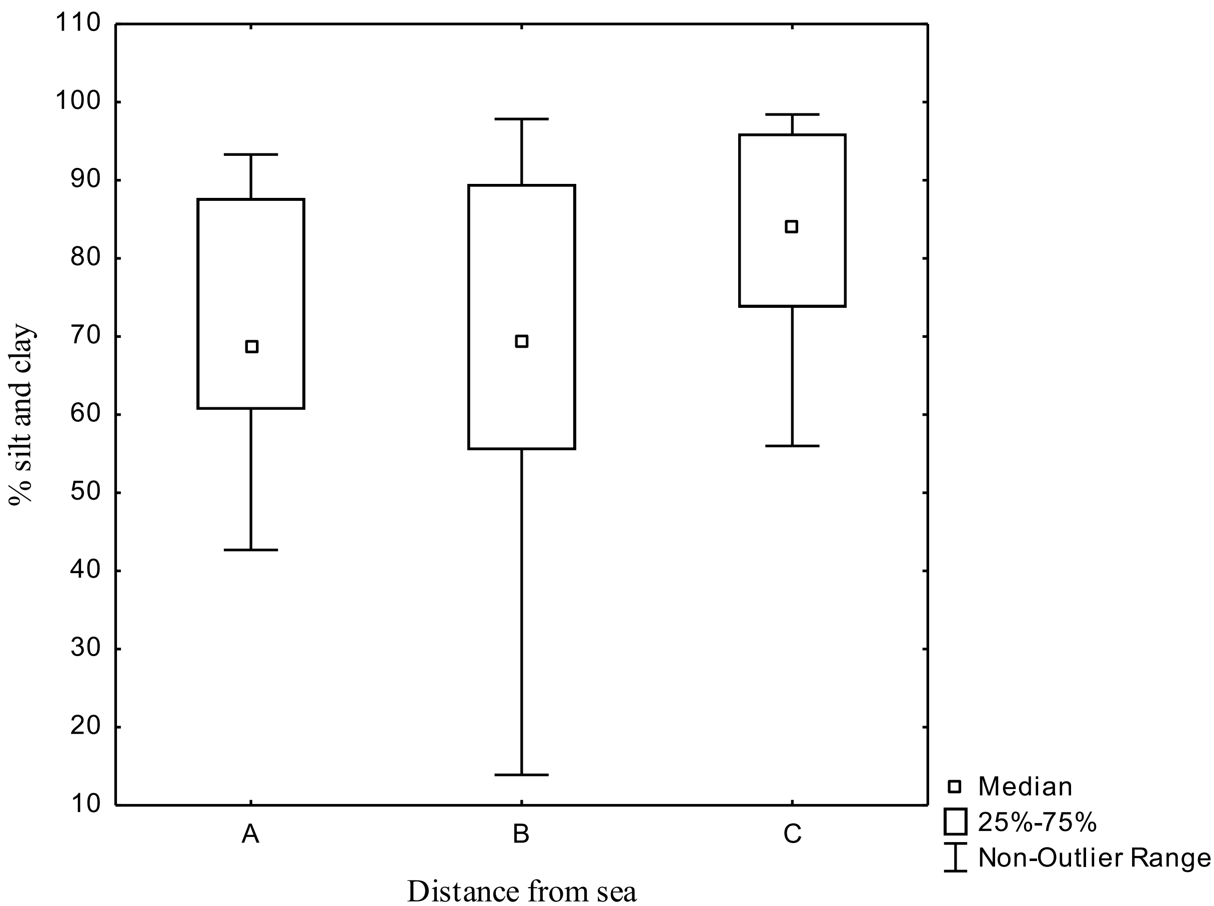


Figure 4.2: Box plot of the percentage silt and clay distribution of the mangrove soils with distance from the sea in Mwache Creek; A-sea-ward, B-mid-forest, C-land-ward.

4.2.2 Bulk density

The average bulk density for the mangrove forest in the creek was 0.88±0.01 g cm^{-3} ranging between 0.70 g cm^{-3} and 1.31 g cm^{-3}. There were significant differences in bulk density among the different sampling sites of the forest (H=2.02; $p<0.05$) but no significant difference ($p>0.05$) among the depth intervals (Table 4.4). Bulk density varied with depth among the different sites (Figure 4.3). Mwakuzimu and Mashazani depicted a similar trend with increasing depth, as well as Bonje A and Ngare. KPA and Bonje B seemed to have a similar trend to the 30 – 50 cm depth interval but varied in the 50 – 100 cm depth interval.

Table 4.4: Mean ± SE (g cm^{-3}) bulk density in the different depth intervals of Mwache sites

Depth interval (cm)	Site						
	Mashazani	**KPA**	**Ngare**	**Mwakuzimu**	**Bonje A**	**Bonje B**	**Average**
0-15	0.75±0.03	0.70±0.01	0.86±0.02	0.90±0.05	1.03±0.01	0.93±0.04	**0.89±0.03**
15-30	0.66±0.06	0.84±0.02	0.91±0.03	0.87±0.06	0.97±0.05	0.93±0.03	**0.88±0.03**
30-50	0.74±0.01	0.83±0.09	0.93±0.02	1.06±0.25	0.98±0.07	0.87±0.05	**0.88±0.03**
50-100	0.69±0.10	0.77±0.01	1.03±0.03	1.00±0.07	1.01±0.05	0.89±0.05	**0.89±0.04**
Average	**0.71±0.03**	**0.79±0.03**	**0.90±0.01**	**0.91±0.05**	**1.00±0.02**	**0.91±0.02**	

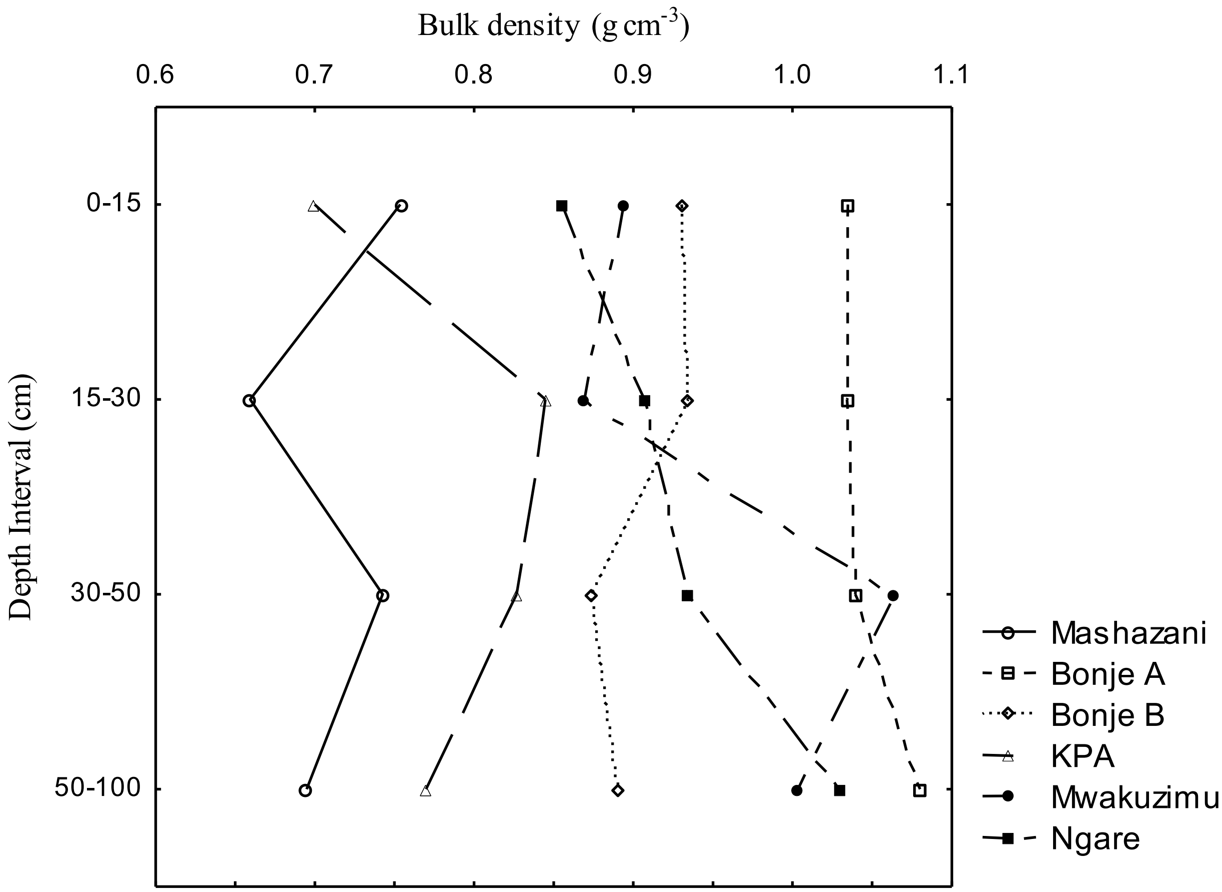


Figure 4.3: Mean bulk density (g cm^{-3}) trends through the soil depth profile in the different sample sites.

Along the sea-land transect, there was a significant difference ($H=14.1$; $p<0.05$) with the bulk density being highest in the land-ward zones (0.97±0.02 g cm^{-3}) and lowest in the mid-tidal zones (0.84±0.02 g cm^{-3}) (Figure 4.4).

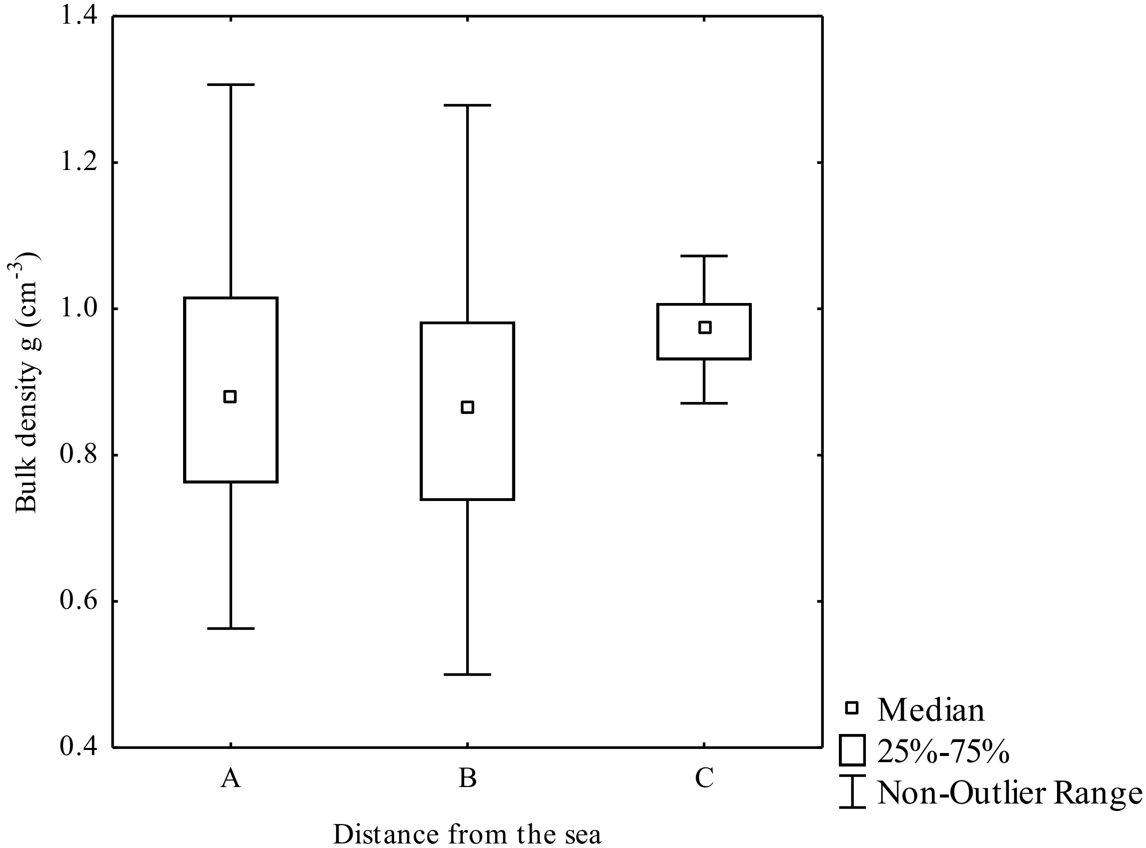


Figure 4.4: Box plot of bulk density of the mangrove soils with distance from the sea in Mwache Creek; A- sea-ward, B-mid-forest, C-land-ward.

4.2.3 Soil organic matter

The percentage soil organic matter in the mangroves of Mwache ranged between 2.1±0.3% in Bonje A and 10.2±0.9 % in Mwakuzimu (average 5.6±0.4 %). It varied significantly among the sites and along transects ($H=65.6$; $p<0.05$). Although not significant ($H=2.56$; $p>0.05$), there was a general increase in organic matter content with an increase in depth (Table 4.5).

Table 4.5: Mean ± SE percentage soil organic matter in different depth intervals in the different sample sites in the study area.

Depth interval	Site						
	Mashazani	KPA	Ngare	Mwakuzimu	Bonje A	Bonje B	Average
0-15	8.64±1.08	3.03±0.75	6.89±0.77	10.49±1.64	1.61±0.39	2.23±0.34	**4.80±0.79**
15-30	10.78±1.87	2.74±0.81	7.85±1.57	9.66±1.75	1.31±0.26	3.68±0.41	**5.41±0.85**
30-50	10.25±0.25	3.00±0.51	8.51±2.13	9.82±2.72	2.27±0.54	4.12±0.81	**5.81±0.84**
50-100	10.50±1.86	3.64±0.27	7.56±1.47	10.83±2.19	3.36±1.00	4.21±0.53	**6.18±0.80**
Average	**10.04±0.66**	**3.10±0.27**	**7.70±0.69**	**10.20±0.91**	**2.14±0.33**	**3.56±0.31**	

Soil organic matter exhibited no significant variations with distance from the sea (H=0.7; p>0.05).The percentage soil organic matter was 6.0±0.7% (range: 1.2 to 14.5%) in the seaward areas, 6.1±0.8 % (range: 1.0 to 14.9%) in the mid-forest and 5.2±0.8% (range: 0.9 to 12.1%) at the land-ward zone (Figure 4.5).

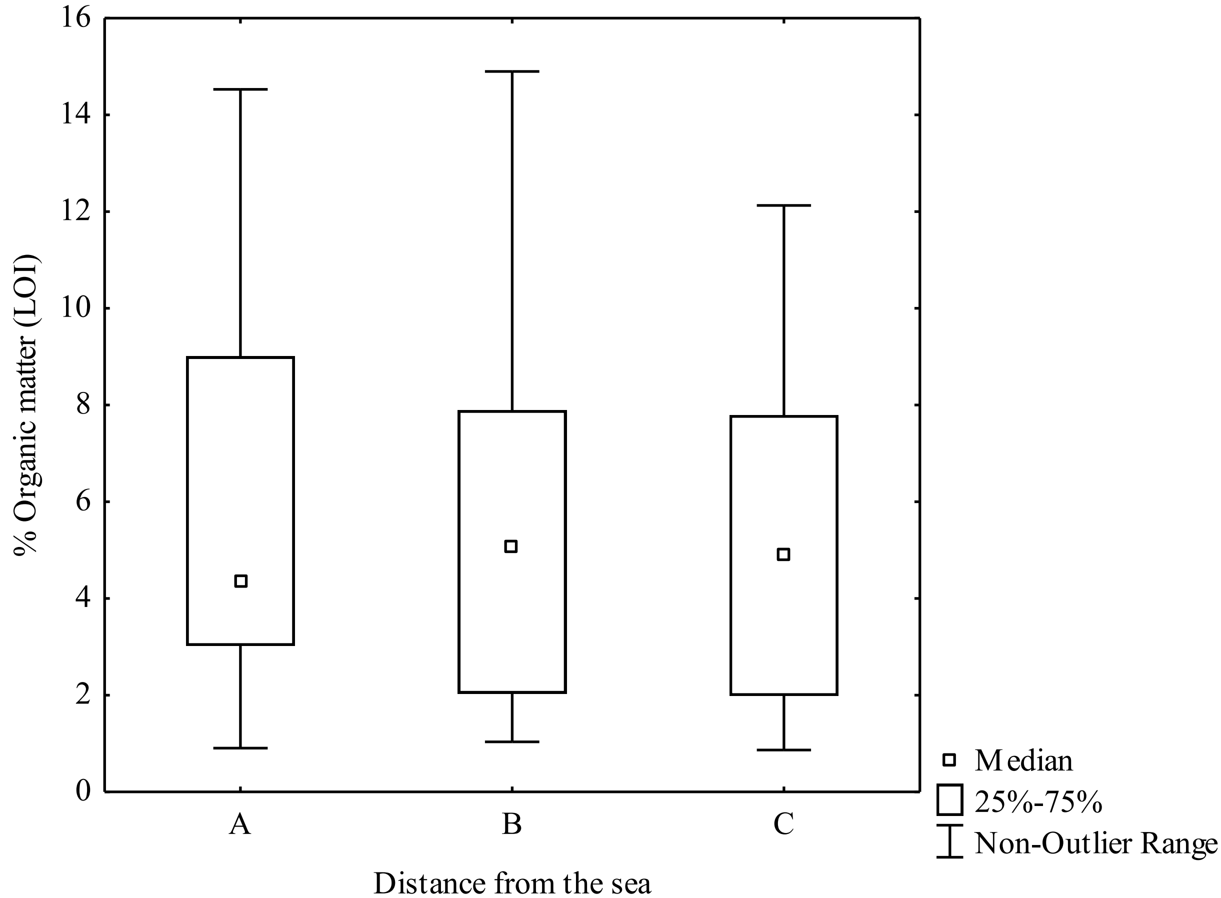


Figure 4.5: Box plot of percentage organic matter with distance from the sea in Mwache Creek; A- sea-ward, B-mid-forest, C-land-ward.

4.2.4 Soil organic carbon concentration (% C)

The mean concentration of soil organic carbon of the sites was 2.0±0.2% C, ranging between 0.4% C in Bonje A and 5.0% C in Mwakuzimu. Overall, there was an increase in C concentration with an increase in soil depth (Figure 4.6). However, the value of C concentration did not differ significantly ($H=4.48$; $p>0.05$) with the depth. Within the sites however, the organic carbon concentration exhibited significant variations (Figure 4.7)

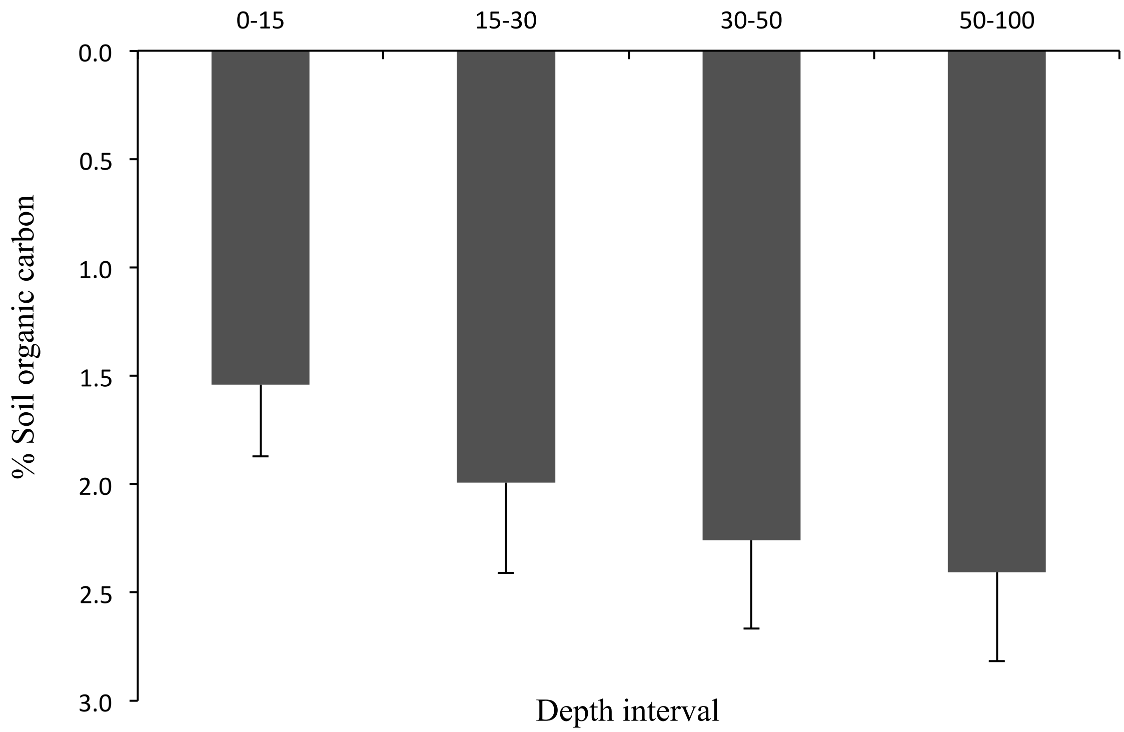


Figure 4.6: Mean + SE soil organic carbon concentration of the mangrove soils in the different depth intervals.

Along the sea-land transects, there was no significant differences in C-concentration ($H=2.51$; $p>0.05$). Generally, the land-ward zones exhibited higher C concentrations than both the mid or seaward zones.

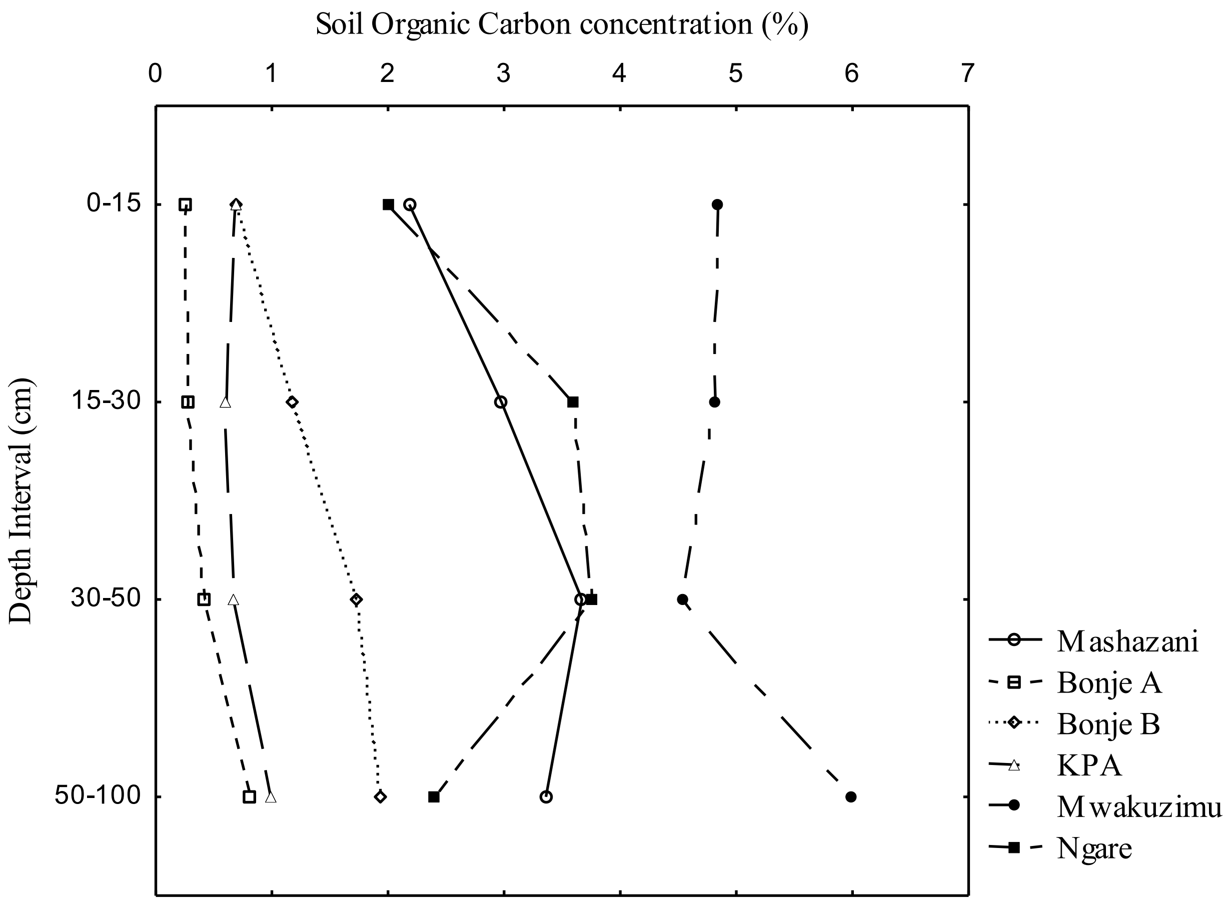


Figure 4.7: Mean soil organic carbon concentration trends through the soil depth profile in the different sampling sites.

It can however be noted in Figure 4.8 that the C concentrations in the 30- 50 and 50-100 depth intervals increased towards mid-forest after which it decreased towards the land-ward areas of the forest.

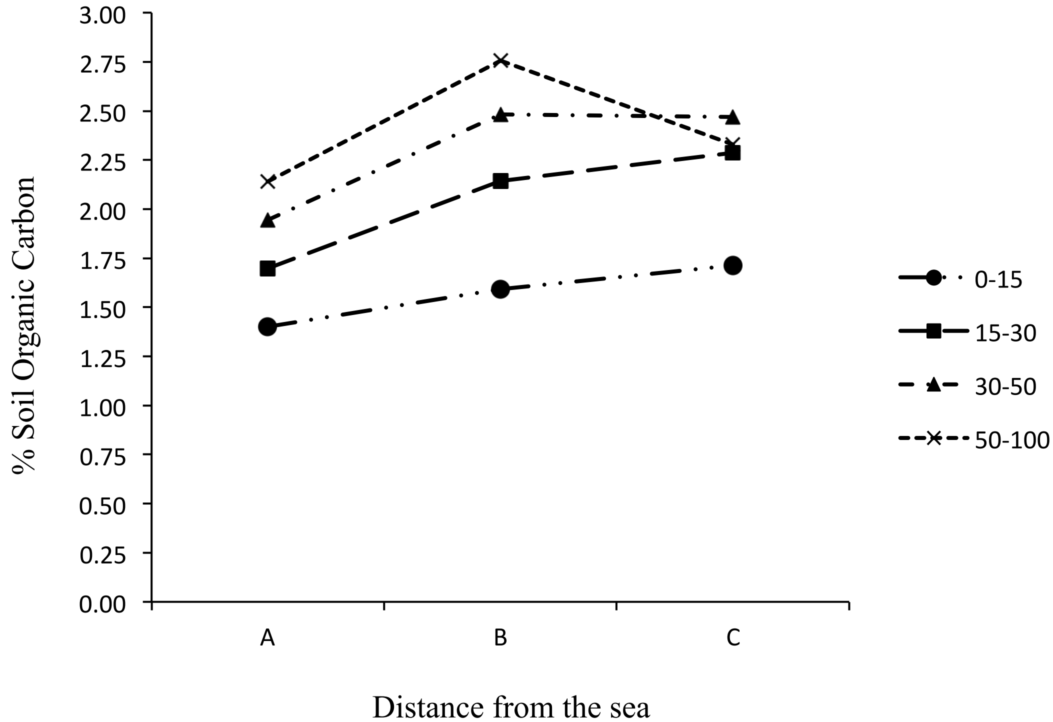


Figure 4.8: Organic carbon concentration of the mangrove soils in A- sea-ward, B-mid-forest, C-land-ward zones of Mwache Creek.

Regression of SOM obtained through LOI against soil organic carbon obtained through the colorimetric method presented a significant relationship (R^2=0.80; $p<0.05$) as illustrated in Figure 4.9. From this, it can be deduced that 43% of the soil organic matter is organic carbon in the soils of the mangrove forest in Mwache.

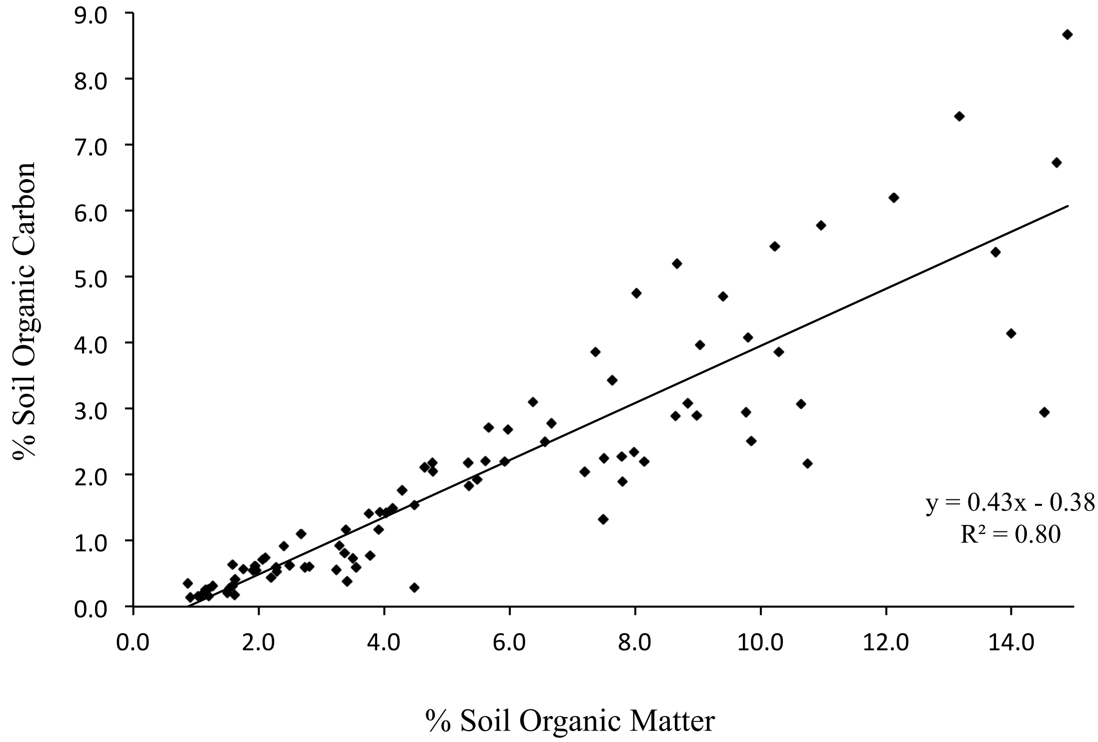


Figure 4.9: Relationship between organic matter and soil organic carbon concentration of soil in Mwache Creek mangrove forest.

4.2.5 Soil organic carbon

The soil organic carbon in Mwache mangrove forest was estimated to be 246.1±71.5 Mg C ha^{-1} (range: 75.8 to 628.3 5 Mg C ha^{-1}). There was a significant difference among sites (H=29.23; $p<0.05$) in SOC with Mwakuzimu recording the highest SOC of 508.26±84.6 Mg C ha^{-1}, followed by Ngare (263.22±51.8 Mg C ha^{-1}), Mashazani (223.44±21.7 Mg C ha^{-1}), Bonje B (191.71±39.0 Mg C ha^{-1}), Bonje A (123.56±10.6 Mg C ha^{-1}) and KPA (78.20±8.4 Mg C ha^{-1}) (Fig 4.10).

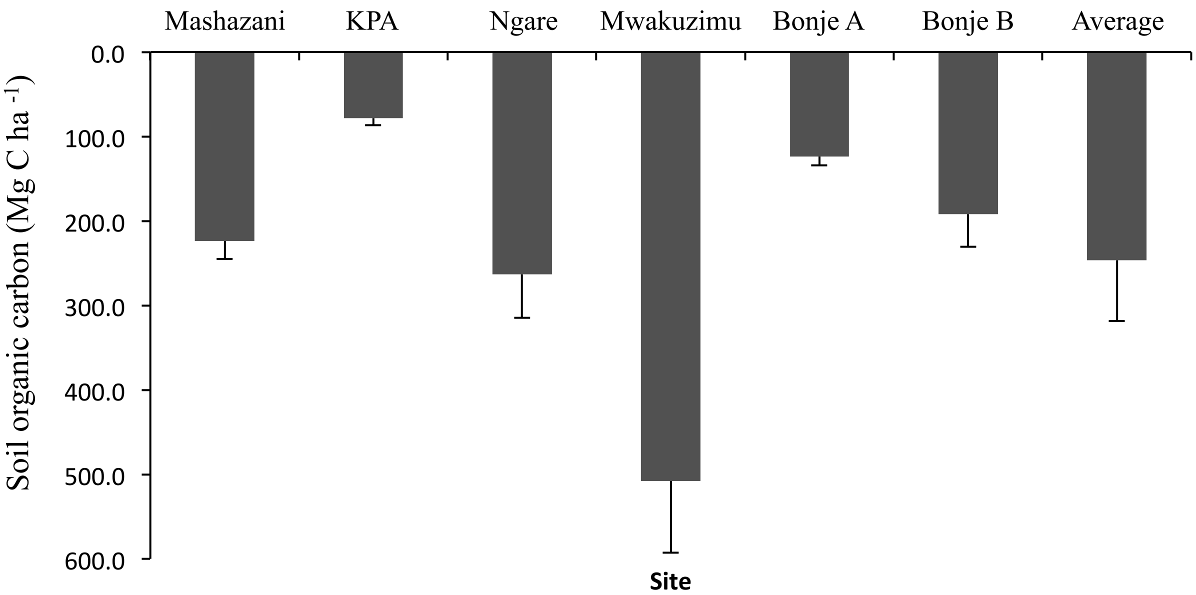


Figure 4.10: Mean +SE SOC among the different sites within the study area.

The 0-15 depth interval had an average of 25.3±3.9 Mg C ha^{-1} (range: 10.4 to 74.2 Mg C ha^{-1}), which increased through to the 50-100 interval to 123.8±20.1 Mg C ha^{-1} (range: 36.3 to 392.2 Mg C ha^{-1}). The observed increase of SOC in depth (Figure 4.11) did display a statistical difference ($p<0.05$; H (3, n=88) =38.82).

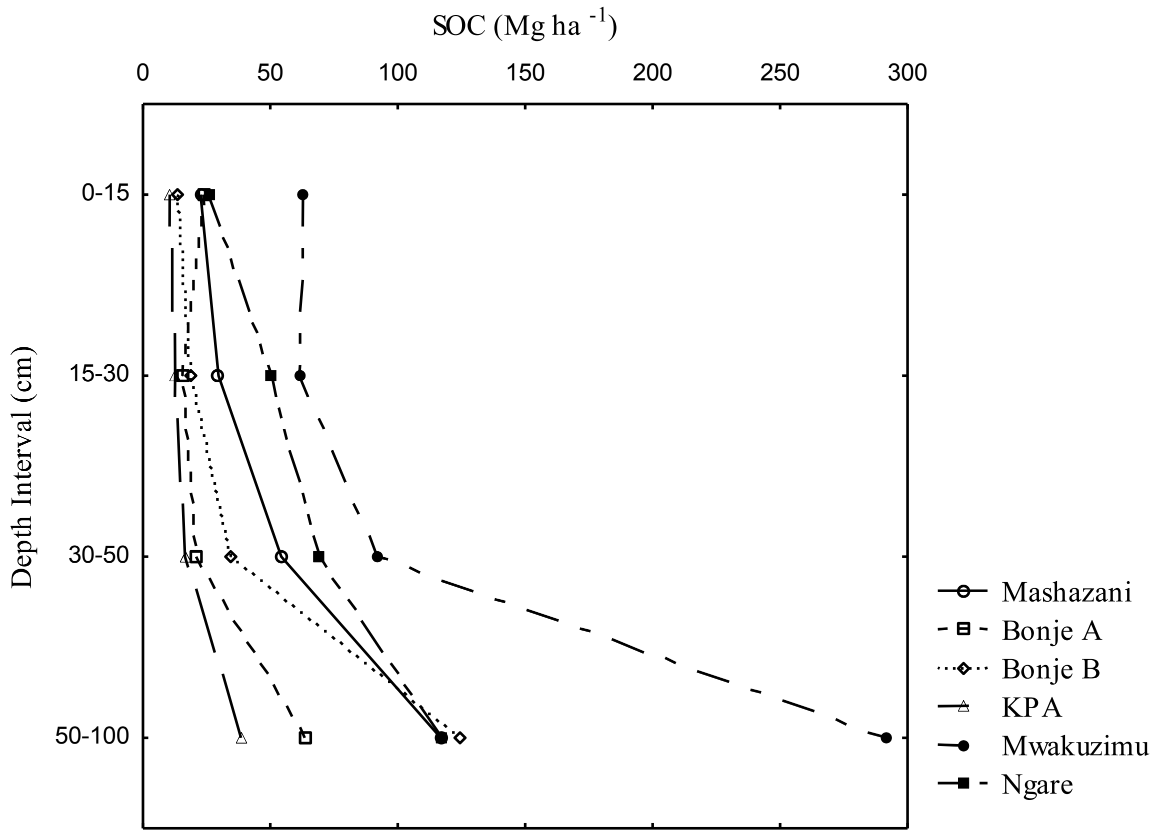


Figure 4.11: Mean soil organic carbon (Mg C ha^{-1}) trends through the soil depth profile in the different sampling sites.

Along transects, the sea-ward zones contained the lowest SOC values compared to the mid-forest and land-ward zones – 49.1, 55.0 and 69.1 Mg C ha^{-1} respectively (Figure 4.12). The four depth intervals had slight variations with increasing distance from the sea as in Figure 4.12 where the 30-50 depth interval had a slight decline towards the mid-forest while the 0-15 depth interval had a decline towards the land-ward zones. The general trend, however, was an increasing SOC value from sea-ward to land-ward zones. No significant difference in SOC was observed with distance from the sea ($H=0.85$; $p>0.05$).

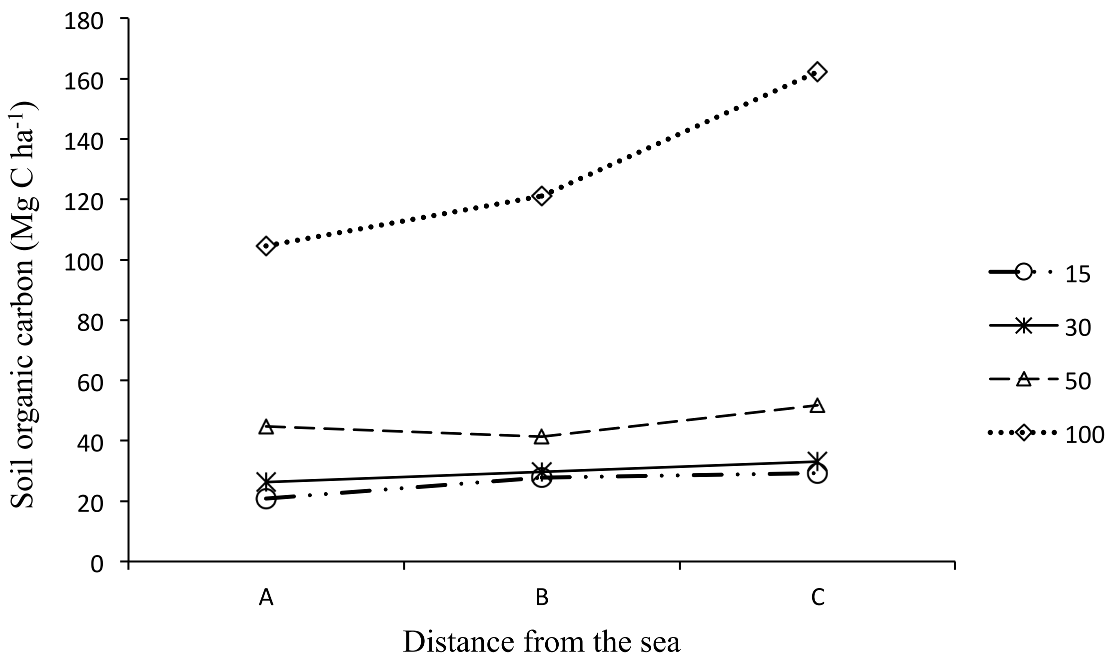


Figure 4.12: Mean soil organic carbon stock trends with depth in the mangroves with distance from the sea in Mwache Creek; A- sea-ward, B-mid-forest, C-land-ward.

4.3 Total Organic Carbon in the mangroves of Mwache Creek

To estimate the ecosystem carbon stock in Mwache, the above and below ground (root and soil) carbon pools were added. On this basis, the average total organic C in the study site was estimated to be 388.9±63.2 Mg C ha^{-1}. Figure 4.13 represents the organic carbon stocks and their allocation in the sampled pools.

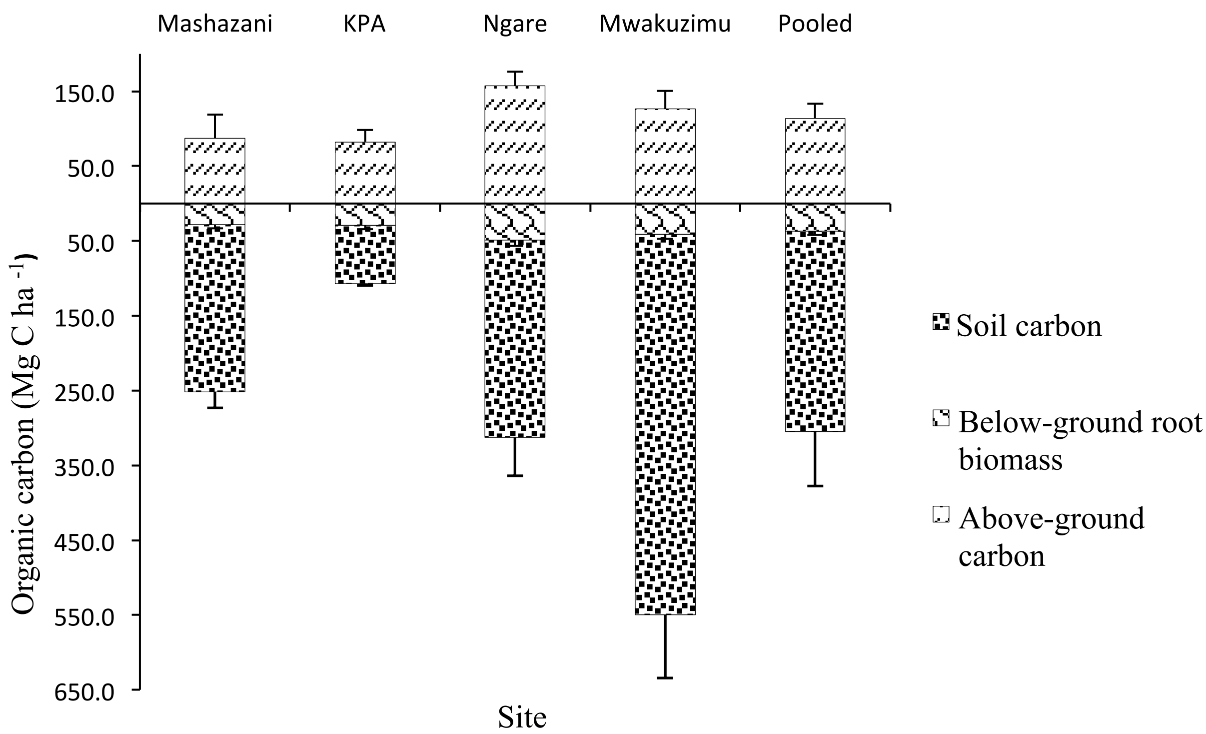


Figure 4.13: Comparison of the distribution of mean+SE of organic carbon stocks in sampled pools in sampled sites in the mangrove forest in Mwache Creek.

4.4 Comparison between highly impacted and less impacted sites

Based on the levels of die-backs observed from the impacts of the 1997/98 and 2006 *El-nino* rains, two transects within the mangrove forest in Mwache Creek were categorized as highly and less impacted; Bonje A and Bonje B, respectively. Significant differences ($p<0.05$) were observed for bulk density, percentage silt and clay, soil organic matter and carbon concentration. Soil bulk density was higher in Bonje A, at 1.00±0.09g cm^{-3}, compared to Bonje B where it was 0.91±0.10 g cm^{-3}. Average percentage silt and clay was higher in Bonje B compared to Bonje A. The average carbon concentration in Bonje B was almost twice that of Bonje A (Figure 4.14); and subsequently the soil organic carbon higher in Bonje B. There was however no significant difference ($p>0.05$) in soil organic carbon stocks between the two sites.

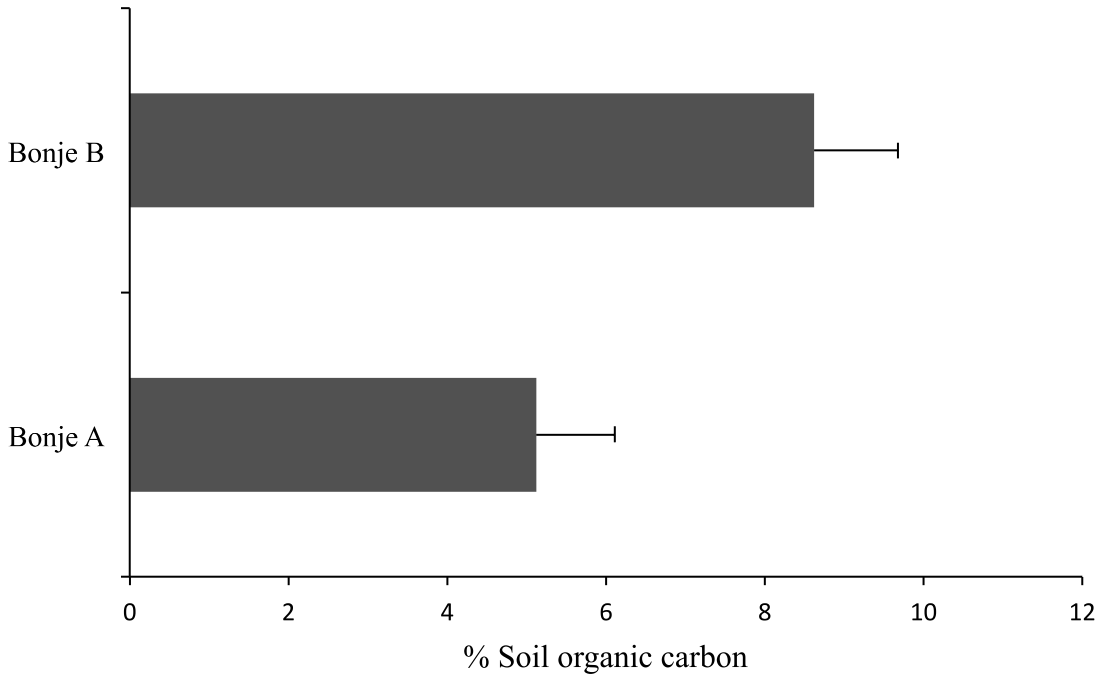


Figure 4.14: Comparison of mean+SE soil organic carbon concentration between the highly degraded (Bonje A) and less degraded (Bonje B) sites in Mwache Creek.

4.5 Estimated emissions from the Mwache mangrove forest

To estimate CO_2 emissions from the Mwache mangrove forest following losses experienced between 1992 and 2009, estimation guidelines from the IPCC 2006 and the 2013 supplement were adopted. Changes in mangrove above ground biomass were estimated using the Stock-Difference method, where differences in the biomass estimated based on percentage changes in cover was used (Table 7). Losses were assumed to have resulted from wood removals and fuel wood removals, and hence the assumption that soil CO_2 emissions were zero was also adopted. It was also assumed that the changes in above ground carbon stocks between 2009 and the study period (2011) were insignificant.

Table 4.6: Estimated CO2 emissions from the mangrove forest in Mwache Creek

Year	**Live biomass change (ha)**	**Estimated C stock loss from live biomass (Mg C ha^{-1})**	**Estimated CO_2 emissions from live biomass loss (Mg CO_2 ha^{-1})**
1992-1994	-17.5 %	34.5	126.4
1994-2000	-17.2%	27.9	102.5
2000-2009	- 20.0%	26.9	98.8
Total (1992-2009)	-45.4%	89.4	328.0

CHAPTER FIVE

DISCUSSION

5.1 Soil properties

5.1.1 Soil structure

Wide variations in soil properties were observed among sites in the mangrove forest of Mwache Creek. Percentage silt and clay varied among the sites and with an increase in depth and also along the intertidal zone. This is a common phenomenon in dynamic coastal systems like mangroves. Tidal current is one of multiple mechanisms for the dispersal and accumulation of sediments in mangrove forests (Kitheka 2000; Adame *et al*., 2010; Tue *et al*., 2011) with the trapped materials originating *in situ* or most commonly transported from other areas and deposited in mangrove areas. A study in the site by Kitheka (2000) reported an increase of grain sizes towards the sea due to tidal action and explained re-suspension during spring tides as the reason for deposition of finer grain sizes with increasing distance from the sea. This was however not the observed trend in the present study as the sea-ward zones exhibited finer grain sizes compared to the mid-tidal zones. This trend may show the inherent capacity of the mangroves in the mid-tidal zone (mostly *Rhizophora mucronata*) to trap more coarse grains because of the mesh-type rooting characteristics. Moreover, high turbidity in the sea-ward and land-ward areas in the mangrove forest has also been documented (Kitheka, 2000). It can be concluded that the sea-ward and land-ward zones experience deposition of sediments from the sea and the hinterlands, respectively. This may be evident from the development of three islands within the creek (one of which is KPA) from accretion of the sediments. The island was colonized by *Sonneratia alba* stands that were absent during 1992 surveys by Ferguson (1993). This is an excellent example of land building by mangroves. Among sites, the less degraded displayed higher percentages of the silt and clay compared to those that were highly degraded. This, and the low silt and clay percentages on the top layers in sites of lower tree density such as Bonje A, could be explained by the level of vegetation cover. The above ground vegetation enhances trapping of sediments (Furukawa and Wolanski,1996; Falconer *et al*., 2001; Van Santen *et al*., 2007; Tue *et al*., 2012) resulting from re-suspension by tides. In addition, the root system efficiently traped particles by slowing the rate of water movement allowing for their settlement (Wolanski, 1995; Young and Harvey, 1996). The sedimentation process is further influenced by the fauna colonizing tree parts (they add to the friction created by the soil and tree surfaces) and the crab burrows (Alongi, 2011). Topography could also affect grain-size distribution by limiting the ability of sediments to settle due to slope influence and by causing re-suspension due to higher velocities from the slope (Adame *et al*., 2010) as depicted by elevated areas in the forest, such as Bonje B and the land-ward areas of Mwakuzimu and Ngare, which demonstrated lower levels of silt and clay.

5.1.2 Bulk density

Bulk density of the soils in Mwache was found to range from 0.7 to 1.0 g cm^{-3} with a mean of 0.88 g cm^{-3} (Table 4.1). These results are comparable to similar studies in Micronesia and Tudor Creek (Kauffman *et al*., 2011; Olagoke, 2012). The values were however higher compared to other values for mangrove soils which ranged from approximately 0.1 to 0.5 g cm^{-3} in Terminos Lagoon in Mexico and Micronesia where mangroves are extensive (Rivera-Monroy *et al.*, 1995; Fujimoto *et al*., 1999; Donato *et al*., 2011). Along the intertidal area, the bulk density inversely related to grain-size distribution; high bulk density corresponded to low percentages of silt and clay as depicted in Figure 5.1 (sandy soil has higher bulk density) (Mehlich, 1972).

In Donato *et al*. (2011) bulk density fluctuated to a depth of one meter and increased thereafter in mangroves across a broad tract of the Indo-Pacific region. On the contrary, Ceron-Breton *et al.* (2010) noted an increase in the bulk density with an increasing depth in their study in Campeche, Mexico. The observed fluctuations in the Mwache Creek mangrove forest, where the bulk density showed no clear trend with increasing depth, may be as a result of the varying vegetation density, the morphology and heterogeneity in the rooting systems of mangroves (also causing varying tidal influences), as well as the fauna presence (Sukardjo, 1994; Castaneda-Moya *et al*., 2011). In addition, Mwache Creek experiences lots of sediment depositions as earlier mentioned (Kitheka *et al*., 2002). This was probably the main reason of the observed unique trends of bulk density in the forest particularly in Bonje A and B which had high bulk density levels, predominantly in the surface and the 50-100 cm layers. These layers could be representations of heavy sediment deposition events in the area.

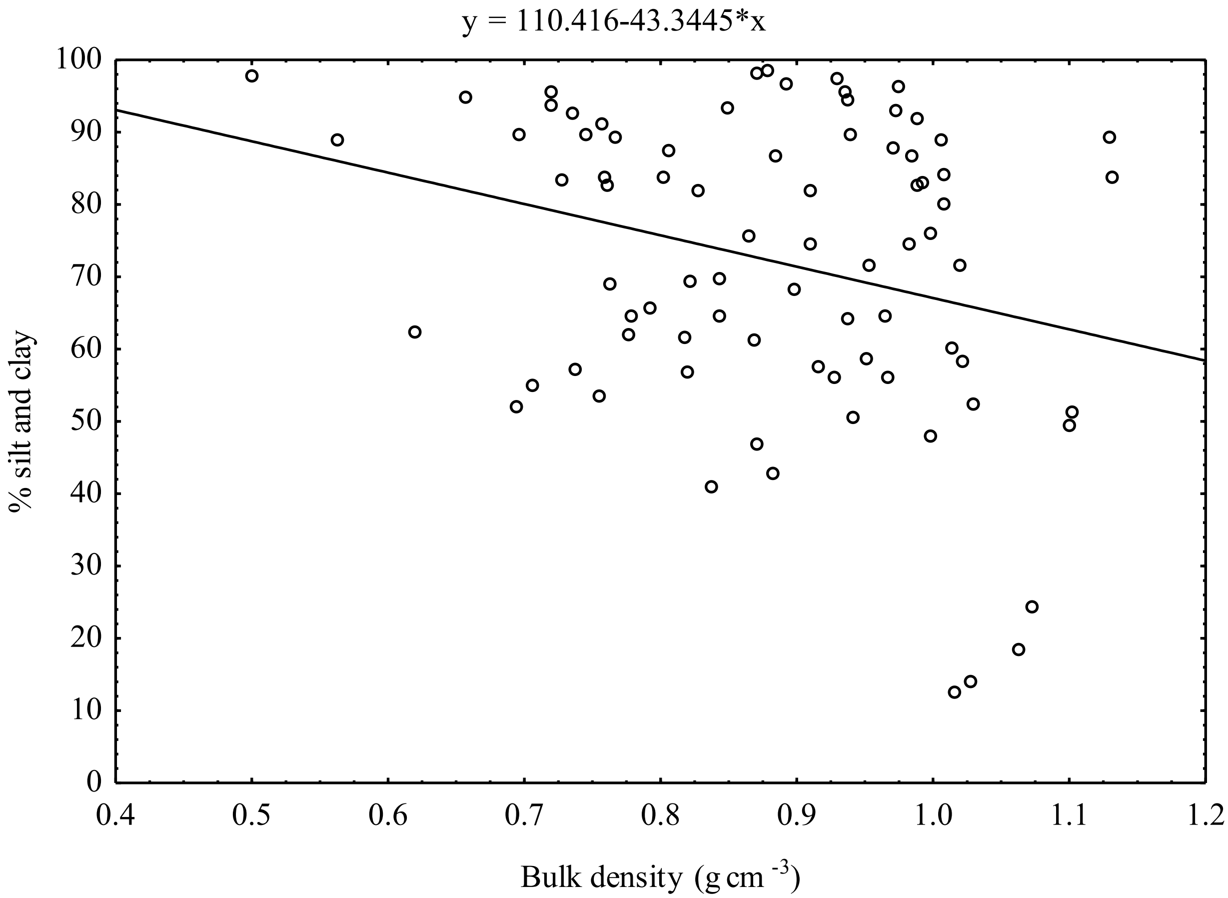


Figure 5.1: Scatter plot of % silt and clay against bulk density in Mwache Creek mangrove forest

High soil organic matter content in mangrove forests is attributable to long periods of tidal flooding and low decomposition rates sustaining anoxic conditions (Ceron-Breton *et al.*, 2010). The percentage organic carbon buried in mangrove forests is highly reliant on environmental conditions (Kjerfve *et al.*, 1999; Kristensen *et al.*, 2008). Mangrove forests and adjacent environments strongly interact and therefore organic material is imported and exported into and from the mangrove ecosystems by tides (Hemminga *et al.*, 1994; Kitheka *et al.*, 2005; Bouillon *et al.*, 2008; Kristensen *et al.*, 2008). In the Mwache Creek mangrove forest the soil organic matter content increased with increasing distance from the sea but decreased from the mid-forest to the land-ward zones as would be expected considering the forest structure. Greater burial of carbon was expected in the sea-ward and land-ward zones due to the prevailing high percentage silt and clay and low bulk density. This however was not the case and may have been attributed by the washing off of the organic matter in the sea-ward zones of mangrove forests (Alongi *et al*., 2005). The forest zonation, where *Avicennia marina* was found in the extreme ends of the intertidal and *R. mucronata* in the mid-forest could also explain the differing trends of the soil organic matter and carbon concentrations. *A. marina* facilitates oxidation of organic matter by leaching oxygen through its roots into the rhizosphere also creating favourable conditions for fauna to make burrows (Lacerda *et al*., 1995; Kristensen *et al*., 2008; Allen *et al*., 2011).The accumulation process of organic matter is enhanced in the mid-forest zone where the drainage may be deficient compared to the sea-ward zone (Santo *et al*., 2011). The low content in the land-ward zones was probably due to the rising salinity further from the sea caused by infrequent tidal inundation (Saintilan, 1997; Kitheka *et al.*, 2002; Bouillon *et al*., 2007). The soil organic carbon concentrations in the depths beyond 30 cm follow a unique trend compared to sub-surface layers; it rises with increasing distance from the sea to mid-forest and decreases thereafter, similar to Palau (Kauffman *et al*., 2011). The unexpected trends in the 0-30 cm intervals, where the soil organic carbon concentrations are highest in the landward zones, give an indication of deposition of organic carbon from extreme events upstream. This may be supported by the observed land use practices in the areas adjacent to the forest such poor tillage practices leading to deposition of sediments into the forest. Variations in the two parameters (OM and OC) with increasing depth were similar in trend to mangroves of Tudor, Indo-Pacific and in Micronesia (Olagoke, 2012; Donato *et al*., 2012; Kauffman *et al*., 2011).

5.2 Total organic carbon

There is higher variability in mangrove carbon stocks around the world (Table 5.1). This factor is attributed to climatic conditions, forest type and age, management conditions and salinity levels (Twilley *et al*., 1992; Kristensen *et al*., 2008). The most productive forests are found in river mediated tropical forests as in South East Asia, Central Africa and Amazon basin. In the Indo-Pacific, Donato *et al*. (2011) estimated mangrove carbon stocks to be 1023 Mg C ha^{-1}. This figure is significantly different from the current study in Mwache creek Kenya, where the average of 388.9±63.2 Mg C ha^{-1} (Range: 189.5 to 676.3 Mg C ha^{-1} was accounted Table 5.1). Total vegetation biomass carbon in the mangrove forest of Mwache creek and the soil organic carbon contributed to 37% and 63% of the total organic carbon stock, respectively. These are consistent with similar studies that have revealed higher carbon stocks in the soils (Boullion *et al*., 2008; Donato *et al*., 2011; Kauffman *et al*., 2011; Olagoke, 2012).

Table 5.1: Comparison of the total carbon stock in Mwache with other mangrove studies.

Study Site	**Total C stocks (Mg C ha^{-1})**	**Source (s)**
Atasta Peninsula	450.0 -3750.0	Santos *et al*., 2011
Australia	94.2	Howe *et al*. 2009
Bohol, Philippines	1456.0 -3707.0	Camacho *et al.,* 2011
China	185.5	Zhong and Qiguo, 2001
Campeche, Mexico	12.0 -222.0	Ceron-Breton *et al,*.2010
Indo-Pacific region	1023.0	Donato *et al*., 2011
Mwache, Kenya	**189.5-676.3**	**This study**
Okinawa, Japan	57.3	Khan *et al*., 2007
Palau, Micronesia	479.0 -1068.0	Kauffman *et al*., 2011
Tabasco, Mexico	472.0 -822.0	Moreno *et al*., 2002
Tudor, Kenya	242.0 – 334.0	Olagoke, 2012
Yap, Micronesia	853.0 -1385.0	Kauffman *et al*., 2011

The live biomass carbon in the mangrove forest varied considerably among the sites. Despite having the highest level of standing biomass, the amount of soil organic carbon in Ngare station was low. The high standing biomass in Ngare could be explained by the fact that the forest is structurally more complex in terms of species richness, canopy height, basal area, and tree density compared to the other sites. Mwakuzimu, on the other hand, had the highest soil organic carbon in the mangrove area though not the highest level of standing biomass carbon. This may have been attributed to burial of organic debris resulting from extraction of mangroves.

Even though it was the only site with all the five species of mangroves recorded in the Mwache Creek, Mwakuzimu demonstrated less structural complexity compared to Ngare, with tree heights spread along various class sizes. The higher structural complexity in Ngare relates to its higher biomass productivity as there is a positive relationship between the two (Jayakody *et al*., 2008). The lower soil organic carbon in Ngare compared to Mwakuzimu is related to the organic matter content and the *A. marina* stem density. *A. marina* negatively influences organic matter content by facilitating for oxidation in the rhizosphere (Lacerda *et al*., 1995; Kristensen *et al*., 2008). Ngare had an *A. marina* cover of almost 30 % compared to that of Mwakuzimu which was 6 %.

Although carrying the highest tree density (2000 trees per hectare), relatively tall trees and higher complexity, KPA contained the least SOC values in the mangrove forest. The over-wash island site is a young monoculture forest (less than 20 years) (Kaino, 2013) and has thus sequestered much less carbon during its existence as carbon burial efficiency increases with the stand age (Kristensen *et al*., 2008). In addition, KPA is an overwash forest of low accretion where most of the nutrients produced are dispersed either to sea or land-ward (Woodroffe, 1985; Jennerjahn and Ittekkot, 2002; Bosire *et al*., 2005; Bouillon *et al*., 2008). The high biomass carbon may be explained by its favorable positioning where there is flooding with every tide cycle (Twilley and Day, 1999) and high productivity due to high litter degradation and autochthonous and allochthonous nutrient recycling (Lee, 1990; Bouillon *et al*., 2002). Mashazani, representing the least stem density (84 trees per hectare), had low total organic carbon compared to the other sites, except for KPA. The higher soil organic carbon in Mashazani compared to that of KPA may be explained by the previous state of the forest when substantial carbon was captured and stored as well as decomposition of debris resulting from extraction. There was found to be the highest percentage silt and clay in Mashazani (Table 4.3) as well as evidence of a once healthier forest from the high stump numbers and poor quality poles (Kaino, 2013). The current structural state and the relatively low values of soil organic carbon in the forest is an indication of loss of previously buried carbon from the area.

5.3 Estimated emissions from Mwache mangroves

Emissions from the mangrove forest at Mwache Creek was estimated at 328 Mg CO_2 per ha. The reduction in the forest cover and consequent CO_2 emissions come at a cost. The social cost of carbon (SCC) is used to estimate the economic damages due to increases in a tonne of CO_2 emissions as well as the value of avoided damages due to emission reduction in a given period of time. Using available estimates, SCC in the study area would range between less than USD 1 to as high as USD 1738/year in the period between 1992 and 2009. The average value from peer-reviewed estimates is USD 43 per tonne of C, translating to USD 214/year in the study site (Yohe *et al*., 2007). This cost is still underestimated considering that the SCC is not comprehensive as it does not include important physical, ecological, and economic impacts of climate change such as ocean acidification, rapid sea level rise, changes in heat and precipitation extremes which have high implications more so on highly vulnerable ecosystems and communities.

CHAPTER SIX

CONCLUSIONS AND RECOMMENDATIONS

6.1 Conclusions

The findings of this study demonstrated the value of the mangrove forest in Mwache Creek with regard to its ability to capture and store vast amounts of carbon most of which is buried in the soils. A significant rise of soil organic carbon stock with increasing depth was established, subjective to the soil organic carbon concentration and bulk density. As for the distribution of the soil organic carbon stocks along the sea-land transects, there was an increase from the sea-ward to the land-ward. It was however deduced that the trend of the stocks along the intertidal zone tended to correspond with the variations in forest structure and complexity. The same was concluded for carbon stock values between highly and less degraded sites in Mwache Creek mangrove forest, which displayed biologically significant differences with the less degraded having higher values. The total organic carbon stock of the forest was found to be 388.92 Mg C ha^{-1}, a value within the range of other mangrove forests globally. 37% of the value was from the live biomass while the remaining 63% was in the soil. The carbon stocks were significantly greater than those of most terrestrial ecosystems and demonstrated the mangrove ecosystems' uniqueness in burying carbon in the soils and to greater depths.

Further, high carbon stocks in the mangrove forest of Mwache Creek point to the likelihood of high carbon emissions from the area in the event of further degradation or deforestation. The forest is degrading at a rate of 2.7 % per annum, greater than the global decline rate of 1-2 % per annum; and higher than the decline rate in the country's mangrove forests of 0.7 % per annum (Kirui *et al*., 2013). Given the aforementioned decline rate the mangrove forest may be reduced by more than 50 % in the next 25 years if the current conditions remain persistent. This would mean carbon emissions of buried carbon, beyond 300 Mg C ha^{-1}, and the destruction of a potentially robust carbon storehouse. Paradoxically, the current degraded state of the forest makes it a good candidate for a REDD+ project based on the potential of additionality to the current carbon stocks. However a REDD+ project would be complicated given the proposed by-pass road construction expected to clear about 20 ha of mangrove, and the damming of R. Mwache which could influence the hydrology of the area.

6.2 Recommendation

Conservation of mangroves is receiving international interests because of their ability to capture and store large stocks of carbon; in addition to providing an array of other ecosystem services. In Kenya, however, historical degradation and transformation of mangroves has impacted on their provision of goods and services to millions of people along the coast and globally who depend directly and indirectly on them. It is important that a holistic approach to mangrove management be adopted for sustainable utilization of mangrove resources and elimination/minimization of adverse socio-economic and environmental effects. The current efforts to develop a national mangrove management plan will assist in harmonizing management and support sustainable use of the mangrove ecosystems. For its effectiveness information on the current and potential use of mangroves is necessary in the planning stages of management planning. The present study provides relevant information which may assist in mangrove resource zonation and management.

The present study has established baseline carbon stocks that can serve as a foundation for future development of a small-scale carbon offset project in Mwache which would serve as a conservation strategy while improving livelihoods of the local community through the derived financial benefits. This would however be of additional value if a framework for a carbon market is developed for the sustainable use of the forest; in which ecological, environmental and economic benefits may be achieved. Steered by relevant authorities and with joint efforts from partners, a REDD+ project could be adopted in this ecosystem which would improve the livelihoods of the locals while greatly preserving the integrity of the forest and the ecosystem services emanating from its existence. This would however require strategies to rehabilitate the degraded areas of the forest and restore its integrity and maximize on its provision of ecosystem services. Further studies however, on the depth extents and the current sediment deposition rate in the unique mangrove forest of Mwache Creek would improve on the knowledge of its carbon capture and burial. Due to the challenges faced by the lack of local factors, there is a research need to establish localized allometric equations, specific wood densities, carbon concentrations and even biomass expansion factors for ease in future carbon assessment studies.

To restore mangrove ecosystems' integrity in the country and regionally, carbon stock assessments in all the mangrove formations should be undertaken to establish the site-specific carbon quantities so as to produce a mangrove carbon map for the country. This would be best achieved if the same considerations would be made for the terrestrial forests and wetlands, which have more coverage in our country compared to mangrove forests. Protection of mangrove forests using economic incentives could lead to reduction in deforestation and degradation which in turn would contribute to the mitigation of climate change. Additionally, given the levels of poverty and illiteracy, this would be among other ways of providing alternative livelihoods and poverty alleviation to reduce the pressure on mangrove forest. Lack of awareness about the value of mangrove ecosystems has further fuelled their rapid destruction. Awareness programs are imperative to increasing knowledge of local people about the ecological and economic values and functions of mangrove forests. The negative impacts of their misuse would in turn enable keen participation of the public in conservation and management of mangroves. Further, the study recommends:

- Accurate assessment of carbon emissions and the implications of loss in carbon stocks to ecosystem services in the mangrove forest of Mwache Creek.
- A study on the impacts of the Mwache mangrove forest degradation on the economic activities (both artisanal and large-scale industries) along the Creek.

CHAPTER SEVEN

REFERENCES

Abuodha P. and Kairo J. (2001). Human-induced stresses on mangrove swamps along Kenya coast. *Hydrobiologia* 458: 255–265.

Adame M.F., Neil D., Wright S. F. and Lovelock C. (2010). Sedimentation within and among mangrove forests along a gradient of geomorphological settings. *Estuarine, Coastal and Shelf Science*. 86:21-30.

Allen D., Dalal R., Rennenberg H. and Schmidt S. (2011). Seasonal variation in nitrous oxide and methane emissions from subtropical estuary and coastal mangrove sediments, Australia. *Plant Biology* 13: 126–133.

Alongi D.M. (2002). Present state and future of the world's mangrove forests. *Environmental Conservation* 29: 331–349.

Alongi D.M. Ramanathan A., Kannan L., Tirendi F., Trott L., Bala Krishna Prasad, M., (2005). Influence of human disturbance on benthic microbial metabolism in the Pichavarum mangroves, VellareColeroon estuarine complex, India. *Marine Biology* 147: 1033-1044.

Alongi D. (2011). Carbon payments for mangrove conservation: ecosystem constraints and uncertainties of sequestration potential. *Environmental Science and Policy* 14:462-470.

Angelsen, A. (2008). *Moving ahead with REDD: Issues, options and implications*. Indonesia. Center for International Forestry Research Jl. CIFOR, Situ Gede, Bogor Barat 16115, Indonesia.

Black K., Byrne K., Mencucci M., Tobin B., Nieuwenhuis M., Reidy B., Bolger T., Saiz G., Green C., Farrell E. and Osborne B. (2009). Carbon stock and stock changes across a Sitks spruce chronosequence on surface-water gley soils. *Forestry* 82(3): 255 – 272.

Bosire J., Dahdouh-Guebas F., Kairo J., Kazungu J., Dehairs F. and Koedam N. (2005). Litter degradation and CN dynamics in reforested mangrove plantations at Gazi Bay, Kenya. *Biological Conservation* 126: 287 – 295.

Bosire J. O., Dahdouh-Guebas F., Kairo J. G., Wartel S., Kazungu J. and Koedam N. (2006). Success rates of recruited tree species and their contribution to the structural development of reforested mangrove stands. *Marine Ecology-Progress Series.* 325: 85-91.

Bosire J., Dahdouh-Guebas F., Walton M., Crona B., Lewis R., Field C. and Koedam N. (2008). Functionality of restored mangroves: A review. *Aquatic Botany* 89:251 – 259.

Bosire J.O., Bandeira S., Rafael J. (2012). *Coastal climate change mitigation and adaptation through REDD+ carbon programs in mangroves in Mozambique: Pilot in the Zambezi Delta.* Determination of carbon stocks through localized allometric equations component. WWF 27p.

Bosire J. O., Kaino J.J., Olagoke A. O., Mwihaki L. M., Ogendi G.M., Kairo J. G., Berger U., Macharia D. (2014). Mangroves in peril: Unprecedented degradation rates of peri-urban mangroves in Kenya. *Biogeosciences*. 11:2623-2634.

Bouillon S., Koedam N., Raman A. and Dehairs F. (2002). Primary producers sustaining macro-invertebrate communities in intertidal mangrove forests. *Oecologia* 130: 441–448.

Bouillon S., Middelburg J., Dehairs F., Borges A., Abril G., Flindt M.,Ulomi S. and Kristensen E. (2007). Importance of intertidal sediment processes and pore water exchange on the water column biogeochemistry in a pristine mangrove creek (RasDege, Tanzania). *Biogeosciences* 4: 311–322 GB2013, doi:10.1029/2007GB003052.

Bouillon S., Borges A., Castaněda-Moya E., Diele K., Dittmar T., Duke N., Kristensen E., Lee S., Marchand C., Middelburg J., Rivera-Monroy V.and Twilley R. (2008). Mangrove production and carbon sinks: A revision of global budget estimates. *Global Biogeochemical Cycles* 22.

Brown S., Sathaye J., Cannell M., and Kauppi P. (1996). Management of forests for mitigation of greenhouse gas emissions. Chapter 24 *In*: Watson R. T., Zinyowera M.C., and Moss R.H. (eds.), *Climate Change 1995: Impacts, Adaptations and Mitigation of Climate Change:Scientific-Technical Analyses. Contribution of Working Group II to the Second Assessment Report of the Intergovernmental Panel on Climate Change*, Cambridge, Cambridge University Press.

Brown S. (1998). Present and future role of forests in global climate change. Pages 59-74 *In*: Gopal B., Pathak P. and Saxena K. *Ecology today: an anthology of contemporary ecological research*. International Scientific Publications, India.

Brown S. (1999). *Land-Use and forestry carbon-offset projects*. USAID Environment Officers Training Workshop. Winrock International Paper.

Brown S., Lim B. and Schlamadinger B. (1999). *Evaluating approaches for estimating net emissions of carbon dioxide from forest harvesting and wood products*. IPCC/OECD/IEA Programme on National Greenhouse Gas Inventories, Paris.

Camacho L., Gevaña D., Carandang A., Camacho S., Combalicer E., Rebugio L. and Youn Y. (2011). Tree biomass and carbon stock of a community managed mangrove forest in Bohol, Philippines. *Forest Science and Technology* 7(4): 161-167.

Ceron-Breton J., Ceron-Breton R., Rangel-Marron M. and Estrella-Cahuich A. (2010). Evaluation of carbon sequestration potential in undisturbed mangrove forest in Terminos Lagoon, Campeche. Pages 295-300 *In*: Mladenov V., Psarris K.,Mastorakis N., Caballero A. and Vachtsevanos G. (eds.) *Development, Energy, Environment, Economics.* International conference on development, energy, environment, economics, Tenerife. WSEAS Press ISSN: 1792-6653.

Charlotte S. and Scholz S. (2006). The role of forests in global climate change: whence we come and where we go. *International Affairs* 82 (5): 861–879.

Chmura G., Anisfeld S., Cahoon D., Lynch J. (2003). Global carbon sequestration in tidal, saline wetland soils. *Global Biogeochemical Cycles* 17 (4): 1111 doi: 10.1029/2002 GB001917.

Costanza R., d'Arge R., de Groot R., Farber S., Grasso M., Hannon B., Limburg K., Naeem S., O'Neill R., Paruelo J., Raskin R., Sutton P. and van den Belt M. (1997). The value of the world's ecosystem services and natural capital. *Nature* 387: 253-260.

Crooks S., Herr D., Laffoley D.,Tamelander J. and Vandever J. (2010). *New Opportunities to Regulate Climate Change through Restoration and Management of Coastal Wetlands and Marine Ecosystems.* World Bank, IUCN, ESA PWA.

Dahdouh-Guebas F., Jayatissa L., Di Nitto D., Bosire J., Lo Seen D. and Koedam N. (2005). How effective were mangroves as a defense against the recent Tsunami? *Current Biological* 15 (12): 443 – 447.

Dahdouh-Guebas F., Mathenge C., Kairo J. and Koedam N. (2000). Utilization of mangrove wood products around Mida Creek (Kenya) among subsistence and commercial users. *Economic Botany* 54: 513–527.

Dahdouh-Guebas F., Van Pottelbergh I., Kairo J., Cannicci S. and Koedam N. (2004). Human-impacted mangroves in Gazi (Kenya): predicting future vegetation based on retrospective remote sensing, social surveys, and distribution of trees. *Marine Ecology Progress Series* 272:77-92.

Danielsen F., Serensen M., Olwig M., Selvam V., Parish F., Burgess N., Hiraishi T., Karunagaran V., Rasmussen M., Hansen L., Quarto A. and Suryadiputra N. (2005). The Asian tsunami: A proactive role for coastal vegetation. *Science* 310: 643.

Dargusch P., Lawrence K., Herbohn J. and Medrilzam C. (2010). A Small-Scale Forestry Perspective on Constraints to Including REDD in International Carbon Markets. *Small-scale Forestry* doi 10.1007/s11842-010-9141-z.

Donato D., Kauffman J., Murdiyarso D., Kurnianto S., Stidham M. and Kanninen M. (2011). Mangroves among the most carbon-rich forests in the tropics. *Nature Geoscience* 4: 293-297.

Donato D., Kauffman J., Mackenzie R., Ainsworth A. and Pfleeger A. (2012). Whole-island carbon stocks in the tropical Pacific: Implications for mangrove conservation and upland restoration. *Journal of Environmental Management* 97: 89-96.

Duarte C., Middelburg J. and Caraco N. (2005) Major role of marine vegetation on the oceanic carbon cycle. *Biogeosciences* 2: 1-8.

Duke N., Ball M. and Ellison J. (1998). Factors influencing biodiversity and distributional gradients in mangroves. *Global Ecology and Biogeography* 7(1): 27 – 47.

Falconer R., Wu Y. and Struve J. (2001). Mathematical modeling of tidal currents in mangrove forests. *Environmental Modeling and Software* 16: 19-29.

FAO (Food and Agricultural Organization of the United Nations) (1994). *Mangrove forest management guidelines*. FAO Forestry Paper 117: 320 Rome, Italy.

FAO (Food and Agricultural Organization of the United Nations) (2006). *Progress towards sustainable forest management*. FAO Forestry Paper 147 Rome, Italy.

FAO (Food and Agricultural Organization of the United Nations) (2007a). *Mangroves of Africa 1980-2005: country reports*. Forest Resources Assessment Programme Working Paper 136 Rome, Italy.

FAO (Food and Agricultural Organization of the United Nations) (2007b). *The World's mangroves 1980-2005.* FAO Forestry Paper 153 Rome, Italy.

FAO (Food and Agricultural Organization of the United Nations) (2009). *Global Forest Resources Assessment 2010, Country Report, Kenya*.

Ferguson W. (1993). *A landscape ecological survey of mangrove resource of Kenya*. Kenya Wildlife Service and Forest Department, Nairobi – Draft.

Field C. (1995). Impact of expected climate change on mangroves. *Hydrobiologia* 295: 75-81.

Fujimoto K., Imaya A., Tabuchi R., Kuramoto S., Utsugi H. and Murofushi T. (1999). Below ground carbon storage of Micronesian mangrove forests. *Ecological Research*, 14, 409-413.

Furukawa K. and Wolanski E. (1996). Sedimentation in mangrove forests. *Mangrove and Salt Marshes* 1: 3–10.

Gibbs H., Brown S., Niles J. and Foley J. (2007). Monitoring and estimating tropical forest carbon stocks: making REDD a reality. *Environmental Research Letter* 2:13.

Gilman E., Ellison J., Duke N. and Field C. (2008). Threats to mangroves from climate change and adaptation options: A review. *Aquatic Botany* 89: 237-250.

Giri C., Zhu Z., Tieszen L.L., Singh A., Gillette S. and Kelmelis J.A. (2008). Mangrove forest distributions and dynamics (1975–2005) of the tsunami-affected region of Asia. *Journal of Biogeography* 35:519–528.

Giri C., Ochieng E., Tieszen L., Zhu Z., Singh A., Loveland T., Masek J. and Duke N. (2011). Status and distribution of mangrove forests of the world using earth observation satellite data. *Global Ecology and Biogeography* 20: 154 – 159.

GoK (Government of Kenya) (2009). *State of the Coast Report: Towards integrated management of coastal and marine resources in Kenya*. National Environment Management Authority (NEMA), Nairobi. 88 pp.

GoK (Government of Kenya) (2010). 2009 *Kenya population and housing census report*.

GoK (Government of Kenya) (2010). *National climate change response strategy*.

Hemminga M., Slim F., Kazungu J., Ganssen G., Nieuwenhuize J. and Kruyt N. (1994). Carbon outwelling from a mangrove forest with adjacent seagrass beds and coral reefs (Gazi Bay, Kenya). *Marine Ecology Program* 106: 291–301.

Herr D. Pidgeon, E. and Laffoley, D. (eds.) (2011). *Blue Carbon Policy Framework: Based on the first workshop of the International Blue Carbon Policy Working Group*. Gland, Switzerland: IUCN and Arlington, USA: CI. vi+39pp.

Houghton R. (2007). Balancing the global carbon budget. *Annual Review of Earth and Planetary Science* 35:313–47. doi 10.1146/annurev.earth.35.031306.140057.

Howe A. J., Rodríguez J.F. and Saco P.M. (2009). Surface evolution and carbon sequestration in disturbed and undisturbed wetland soils of the Hunter estuary, southeast Australia. *Estuarine, Coastal and Shelf Scienc*e 84:75-86.

IPCC (2003). *Good Practice Guidance for Land Use, land-Use Change and Forestry*. Penman J., Gytarsky M., Hiraishi T., Kruger D., Pipatti R., Buendia L., Miwa K., Ngara T., Tanabe K. and Wagner F. (Eds). Intergovernmental Panel on Climate Change (IPCC), IPCC/IGES, Hayama, Japan.

IPCC (2006). *2006 IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme*, Eggleston H.S., Buendia L., Miwa K., Ngara T. and Tanabe K. (eds). Published: IGES, Japan.

IPCC (2007). Intergovernmental Panel on Climate Change. Climate Change: The Physical Basis. *In*: *Contribution of Working Group 1 to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press.

IPCC (2014). *2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands*, Hiraishi T., Krug T., Tanabe K., Srivastava N., Baasansuren J., Fukuda M. and Troxler T.G. (eds). Published: IPCC, Switzerland.

IUCN (2006). *Ecological and socio-economic values of Mangrove ecosystems in Tsunami affected areas: Rapid ecological-economic-livelihood assessment of Ban Naca and Ban Bangman in Ranong Province, Thailand*.

Jayakody J., Amarasinghe M., Pahalawattaarachchiand V. and De Silva K. (2008). Vegetation structure and potential gross primary productivity of mangroves at Kadolkele in Meegamuwa (Negombo) estuary, Sri Lanka. *Sri Lanka Journal of Aquatic Science* 13: 95-108.

Jennerjahn T. and Ittekkot V. (2002). Relevance of mangroves for the production and deposition of organic matter along tropical continental margins. *Naturwissenschaften* 89: 23–30.

Kaino J. (2013). *Status and recovery patterns of mangrove forest at Mwache creek, Mombasa after the El nino event of 1997/98 and 2006.* MSc. Thesis submitted to Egerton University – unpublished.

Kairo J., Dahdouh-Guebas F., Bosire J. and Koedam N. (2001). Restoration and management of mangrove systems — a lesson for and from the East African region. *South African Journal of Botany* 67: 383–389.

Kairo J., Bosire J. and Mohamed, M. (2005). *Assessment of the effects of oil spill on the mangrove forests of Port Reitz, Mombasa*. Kenya Marine and Fisheries Research Institute. Mangrove System Information Service, Mombasa.

Kairo J. G. (2006). *Structural Development and Productivity of Replanted Mangrove Plantations in Kenya.* Alcoa Practitioner Fellowship Program. Final Technical Report. WWF-US, Institute of International Education, Washington D.C. 26pp.

Kathiresan K. and Bingham B. (2001). Biology of mangroves and mangrove ecosystems. *Advances in Marine Biology* 40: 81–251.

Kauffman J., Heider C., Cole T., Dwire K.A. and Donato D.C. (2011). Ecosystem C Pools of Micronesian Mangrove Forests: Implications of land use and climate change. *Wetlands* 31:343–352. DOI 10.1007/s13157-011-0148-9.

Kauffman J. and Donato D. (2012). *Protocols for the measurement, monitoring and reporting of structure, biomass, and carbon stocks in mangrove forests*. Working Paper 86.CIFOR, Bogor, Indonesia.

Khan M.N.I., Suwa R. and Hagihara A. (2007). Carbon and nitrogen pools in a mangrove stand of Kandelia obovata (S., L.) Yong: vertical distribution in the soil-vegetation system. *Wetlands Ecology and Management* 15: 141–153.

Kirui B., Kairo J., Bosire J., Viergever K., Rudra S., Huxham M. and Briers R. (2013). Mapping of mangrove forest land cover change along the Kenya coastline using Landsat imagery. *Ocean & Coastal Management*, 10.1016/j.ocecoaman.2011.12.0 ISSN: 09645691.

Kitheka J. (2000). *Suspended sediment transport and exchange in Port Reitz creek with special focus on the mangrove fringed Mwache wetland, Kenya*. MSc. Thesis submitted to University of Nairobi.

Kitheka J., Ongwenyi G. and Mavuti K. (2002). Dynamics of suspended sediment exchange and transport in a degraded mangrove creek in Kenya. *Ambio* 37(7-8): 580 – 587.

Kitheka J., Ongwenyi G. and Mavuti K. (2003). Fluxes and exchange of suspended sediments in tidal inlets draining a degraded mangrove forest in Kenya. *Estuarine, Coastal Shelf Science* 56: 655 - 667.

Kitheka J., Obiero M. and Nthenge P. (2005). River discharge, sediment transport and exchange in the Tana Estuary, Kenya. *Estuarine, Coastal Shelf Science* 63: 455 - 468.

Kjerfve B., Drude de Lacerda L., Rezende C. and Coelho-Ovalle A (1999). Hydrological and hydrogeochemical variations in mangrove ecosystems, pages 71-82. *In*: A. Yáñez–Arancibia y A. L. Lara–Domínquez (eds.). *Ecosistemas de Manglar en América*

*Tropical.*Instituto de Ecología A.C. México, UICN/ORMA, Costa Rica, NOAA/NMFS Silver Spring MD USA. 380 p.

Kokwaro J. (1985). The distribution and economic importance of the mangrove forests of Kenya. *Journal of the East African Natural History Society* 75(188): 1–12.

Komiyama A., Poungparn S. and Kato S. (2005). Common allometric equations for estimating the tree weight of mangroves. *Journal of Tropical Ecology* 21:471-477.

Komiyama A., Ongino J. and Poungparn S. (2008). Allometry, biomass, and productivity of mangrove forests: A review. *Aquatic Botany*, 89: 128-137.

Kristensen E., Bouillon S., Dittmar T. and Marchand C. (2008). Organic carbon dynamics in mangrove ecosystems. *Aquatic Botany* 89: 201-219.

Lacerda L.D., Ittekkot V. and Patchineelam S. R. (1995). Biogeochemistry of mangrove soil organic matter: a Comparison between *Rhizophora* and *Avicennia* soils in South-eastern Brazil. *Estuarine, Coastal and Shelf Science* 40, 713–720.

Laffoley D. and Grimsditch G. (2009). *The management of natural coastal carbon sinks*. A short summary. IUCN, Gland, Switzerland. 8 pp.

Lasco R., Pulhin F., Sanchez P., Villamor G. and Villegas K. (2008). climate change and forest ecosystems in the Philippines: vulnerability, adaptation and mitigation. *Journal of Environmental Science and Management* 11(1):1-14.

Lee S. (1990). Primary productivity and particulate organic matter flow in an estuarine mangrove-wetland in Hong Kong. *Marine Biology* 106: 453–463.

Lovelock C. and Ellison J. (2007). Vulnerability of mangroves and tidal wetlands of the Great Barrier Reef to climate change, Pages 237–269. *In*: Johnson J. and Marshall P. (Eds.), *Climate Change and the Great Barrier Reef: A Vulnerability Assessment. Great Barrier Reef Marine Park Authority and Australian Greenhouse Office,* Australia.

Lugo A. E. and S. C. Snedaker. (1974). The ecology of mangroves. *Annual Reviews of Ecology and Systematics* 5:39-64 DOI: 10.1146/annurev.es.05.110174.000351

McLeod E. and Salm R. (2006). *Managing Mangroves for Resilience to Climate Change*. IUCN, Gland, Switzerland. 64pp.

Mehlich A. (1972). Uniformity of expressing for calculating results on a volume basis. *Plant Analysis* 3(5): 417-424.

Mitra A., Sengupta K. and Banerjee K. (2011). Standing biomass and carbon storage of above-ground structures in dominant mangrove trees in the Sundarbans. *Forest Ecology and Management* 261:1325–1335.

Moreno E., Pena A., Castorena C., Solorio C. and Lopez D. (2002). Mangrove of Tabasco, a natural carbon reservoir. *Madera y Bosques*. Special issue :115-128.

Nellemann C., Corcoran E., Durate C., Valdes L., DeYoung C., Fonseca L. and Grimditch G. (2009). *Blue Carbon. The Role of Healthy Oceans in Binding Carbon.* United Nations Environment Program. GRID-Arendal, ISBN: 978-82-7701-060-1 www.grida.no

Nguyen H. (2005). *Mangrove degradation: causal chain analysis*. Training Course on Sustainable Management of Mangrove Ecosystems for the UNEP-GEF-South China Sea Project on Reversing Marine Environmental Degradation Trends in the South China Sea and the Gulf of Thailand

Ochiewo J. (2001). Socio-economic aspects of water management along the coast of Kenya. *Hydrobiologia* 458: 267-273.

Olagoke A. O. (2012). *Vegetation dynamics and carbon distribution in a heavily impacted peri-urban mangrove forest in Tudor Creek, Kenya*. MSc Thesis submitted to the Institute of International Forestry and Forest Products, Technische Universitaet Dresden, Germany 82pp

Parker C., Mitchell A., Trivedi M., Mardas N. and Sosis K. (2009). *The Little REDD+ Book*. Global Canopy Foundation. Global Canopy Programme John Krebs Field Station Oxford OX2 8QJ UK.

Pendleton L, Donato D., Murray B., Crooks S. and Jenkins W. (2012). Estimating Global ''Blue Carbon'' Emissions from Conversion and Degradation of Vegetated Coastal Ecosystems. *PLoS ONE* 7(9): e43542. doi:10.1371/journal.pone.0043542.

Primavera J. (1997). Socioeconomic impacts of shrimp culture. *Aquaculture Resources* 28: 815-827.

Primavera J. (2005). Mangroves, fishponds and the quest for sustainability. *Science* 310:57-59.

Ray R., Ganguly D., Chowdhury C., Dey M., Das S., Dutta M., Mandal S., Majumder N., De T., Mukhopadhyay S. and Jana T. (2011). Carbon sequestration and annual increase of carbon stock in a mangrove forest. *Atmospheric Environment.*45(28): 5016-5024.

REDD Readiness Progress Fact Sheet Kenya, March (2013**)**. Accessed on: 18/04/2013 via, www.forestcarbonpartnership.org/sites

Rivera-Monroy V., Twilley R., Boustany R., Day J., Vera-Herrera F. and Ramirez M. (1995). Direct dentrification in mangrove sediments in Terminos Lagoon, Mexico. *Marine Ecology Progress Series* 126: 97-109.

Saenger P. and Snedaker S.C. (1993). Pantropical trends in mangrove above-ground biomass and annual litterfall. *Oecologia* 96: 293-299.

Saintilan N. (1997). Above- and below-ground biomasses of two species of mangrove on the Hawkesbury River estuary, New South Wales. *Marine Freshwater Resources* 48: 147-152.

Santos G., Ceron-Breton R., Ceron-Breton J., Sanchez J., Hernandez D., Muriel M. and Chavez A. (2011). Preliminary study of carbon storage rate in mangrove soils in Atasta

Peninsula, Campeche, México. *Recent Advances in Fluid Mechanics, Heat & Mass Transfer and Biology* 111-116.

Scholz I. and Schmidt L. (2008). *Reducing Emissions from Deforestation and Forest* Degradation in Developing Countries: Meeting the main challenges ahead. Briefing Paper.

Siikimaki J., Sanchirico J. and Jardine S. (2012). Global economic potential for reducing carbon dioxide emissions from mangrove loss. *PNAS* Early Edition: 1-6.

Siikimaki J., Sanchirico J., Jardine S., McLaughlin D. and Morris D. (2012). *Blue Carbon; Global options for reducing emissions from the degradation and development of coastal ecosystems.* Resources for the Future.

Spalding M., Blasco F. and Field C. (eds) (1997). *World mangrove atlas*. The International Society for Mangrove Ecosystems, Okinawa.

Spalding M., Kainuma M. and Collins L. (eds) (2010).*World Atlas of Mangroves*. Earthscan, London.

Sukardjo S. (1994). Soils in the mangrove forests of the Apar Nature Reserve, Tanah Grogot, East Kalimantan, Indonesia. *Southeast Asian studies* 32(3): 385-398.

Tamooh F., Huxham M., Karachi M., Mencuccini M., Kairo J. and Kirui B. (2008). Below-ground root yield and distribution in natural and replanted mangrove forests at Gazi bay, Kenya. *Forest Ecology and Management* 256: 1290–1297.

Tomlinson P. (1986). *The botany of mangroves*. Press Syndicate of the University of Cambridge. The Pitt Building. Trumpington Street, Cambridge CB2 IRP 40 West 20th Street, New York, NY 10022-4211, USA 10 Stamford Road, OAkleig, Melbourne 3166, Australia. Cambridge University Press.

Tue N., Hamaoka H., Sogabe A., Quy T., Nhuan M. and Omori K. (2011). Sources of Sedimentary Organic Carbon in Mangrove Ecosystems from Ba Lat Estuary, Red River, Vietnam pages 151–157. In: Omori K., Guo X.,Yoshie N., Fujii N., Handoh I., Isobe A. and Tanabe S. (Eds) *Marine Environmental Modeling & Analysis*.

Twilley R. and Day J. (1999). The productivity and nutrient cycling of mangrove ecosystems, page 127-152.*In*: Yáñez-Arancibia A. and Lara-Domínguez A. (eds.). *Ecosistemas de Manglar en América* Tropical.Instituto de Ecología A.C. México, UICN/ORMA, Costa Rica, NOAA/NMFS Silver Spring MD USA. 380 p.

Twilley R., Chen R. and Hargis T. (1992). Carbon sinks in mangroves and their implications to carbon budget of tropical coastal ecosystems. *Water Air Soil Pollution*. 64: 265-288.

UNEP (2003). *Mangroves of East Africa*. UNEP-WCMC Biodiversity Series No. 13. Available at http://www.unepwcmc. org/resources/publications/ss1/WCMCMangroves.

UNEP-WCMC (2006). *In the Front Line: Shoreline Protection and Other Ecosystem Services from Mangroves and Coral Reefs*. UNEP-WCMC, Cambridge, 33 pp.

UNEP (2007). *Mangroves of Western and Central Africa*. UNEP-Regional Seas Programme/UNEP-WCMC. http://www.unep-wcmc.org/resources/publications/ UNEP_WCMC_bio_series/26.htm

UNEP (2012). Mangroves in the WIO region. *The Seventh Conference of the Parties to the Convention for the Protection, Management and Development of the Marine and Coastal Environment of the Western Indian Ocean (Nairobi Convention)* UNEP(DEPI)/EAF/CP.7/Inf15a

UNFCCC (2008). *United Nations Framework Convention on Climate Change 2008*. http://unfccc.int.

Valiela I., Bowen J. and York J. (2001). Mangrove Forests: One of the World's Threatened Major Tropical Environments *BioScience* 51(10):807.

Van Santen P., Augustinus P., Janssen-Stelder B., Quartel S. and Tri N. (2007). Sedimentation in an estuarine mangrove system. *Journal of Asian Earth Science* 29:566–575.

Wass P. (ed.) (1995). *Kenya's Indigenous Forests: Status, Management and Conservation*. IUCN, Grand, Switzerland, and Cambridge, UK.

Winrock International (1999). *Field tests of carbon monitoring methods in forestry projects*.

Wolanski E. (1995). Transport of sediment in mangrove swamps. *Hydrobiologia* 295: 31-42.

Woodroffe C. (1985). Studies of a mangrove basin, Tuff Crater, New Zealand: I. Mangrove biomass and production of detritus. *Estuarine and Coastal Shelf Science* 20: 265–280.

Yohe, G., Lasco R., Ahmad Q., Arnell N., Cohen S., Hope C., Janetos A. and Perez R. (2007). Perspectives on climate change and sustainability. In: Parry M., Canziani O., Palutikof J., van der Linden P. and Hanson C. (Eds.) *Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, Cambridge University Press, Cambridge, UK, 811-841.

Young B. and Harvey E. (1996). A spatial analysis of the relationship between mangrove (*Avicennia marina* var. *australasica*) Physiognomy and Sediment Accretion in the Hauraki Plains, New Zealand. *Estuarine and Coastal Shelf Science* 42: 231–246.

Zhong L. and Qiguo Z. (2001). Organic carbon content and distribution in soils under different land uses in tropical and subtropical China. *Plant and Soil* 231:175-185.

APPENDICES

List of publications

Bosire J., Kaino J., Olagoke A., Mwihaki L., Ogendi G., Kairo J., Berger U., and Macharia D. (2014) Mangroves in peril: unprecedented degradation rates of peri-urban mangroves in Kenya. *Biogeosciences* 11: 2623 – 2634.